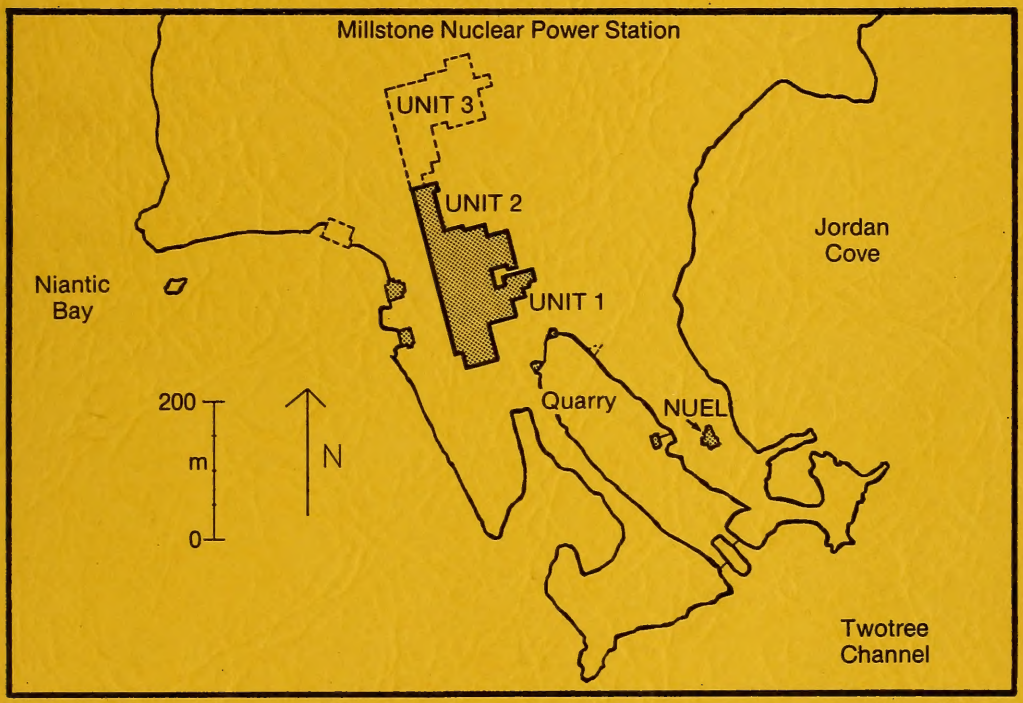
Monitoring the Marine Environment of Long Island Sound at Millstone Nuclear Power Station

Waterford, Connecticut



SUMMARY OF STUDIES PRIOR TO UNIT 3 OPERATION

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Contents

SUMMARY 1
ROCKY INTERTIDAL STUDIES 1
BENTHIC INFAUNA 3
LOBSTER POPULATION DYNAMICS 6
EXPOSURE PANEL PROGRAM 8
FISH ECOLOGY 9
WINTER FLOUNDER STUDIES 11
ACKNOWLEDGEMENTS 15

SUMMARY

The Millstone Nuclear Power Station (MNPS) is located on the north shore of Long Island Sound (LIS) in Waterford, Connecticut. The station consists of three operating units: Millstone Unit 1 commenced commercial operation in 1970, Unit 2 in 1975 and Unit 3 in 1986.

Extensive studies of the potential impacts of MNPS on local marine flora and fauna have been conducted since 1968. During this period studies have consistently been reviewed and updated to assure that the best available methods were used. This report summarizes data from all monitoring programs performed at MNPS during 2-unit operation and is intended to provide information which will establish the baseline for all impact assessment studies during 3-unit operating conditions.

ROCKY INTERTIDAL STUDIES

Attached plant and animal species on local rocky shores were identified, temporal and spatial patterns of occurrence and abundance of these benthic species were examined, and physical and biological factors that induce variability were identified. Qualitative algal collections, quantitative studies of intertidal organisms, recolonization studies, and *Ascophyllum nodosum* studies were performed to assess impact and are summarized below.

The local flora, as characterized by qualitative algal collections, has shown consistent spatial and temporal patterns of distribution during Unit 1 and 2 operation. Overall, 158 algal species have been identified since the inception of the monitoring program in 1979, consisting of 73 reds, 40 browns, and 45 greens. Several of these species were site-specific, and others were seasonally important. The greatest number of algal species recorded from a single collection usually occurred in spring-early summer for all stations. The most species collected in any month was 117 in July and the fewest (101) in March. The most species collected at any station was 131 at Bay Point and the least was 109 at both Seaside Exposed and Twotree Island. Yearly, the greatest species number generally occurred at White Point.

The proportions of reds, browns, and greens are similar throughout the area and independent of species number when analyzed by month and station. Annual percentages of species in each division

(1979-1985) ranged from 44-47% for reds, browns 25-27%, and greens 26-30%. The divisional proportions vary seasonally: reds are more prevalent in August-December, browns in February-May, and greens in May-July. When represented as percent occurrence of reds, browns, and greens (46:25:29), the local flora is consistent with those reported by other researchers in the northwest Atlantic.

Quantitative studies show intertidal zonation patterns typical of rocky shores throughout New England, with the high intertidal dominated by barnacles, the mid intertidal by barnacles and fucoids, and the low intertidal dominated by *Chondrus crispus*, a perennial red alga. The abundances of these major components of local rocky shore communities vary over time and space. Variations are predictable and explainable in terms of seasonality, degree of exposure, intertidal height, inter- and intraspecific competition, and life-history of the organisms. Changes to communities have been minor, indicating stable environmental conditions during 2-unit operation.

An exception to the local stability is the development of a community dominated by opportunistic ephemeral algae after the opening of the second quarry cut in August 1983 at Fox Island-Exposed (FE), the station closest to the discharges. This change was attributed to thermal incursion and water temperatures in excess of 28 °C. High water temperatures in late summer 1984 were responsible for the elimination of the perennial algae *Chondrus crispus*, *Fucus vesiculosus* and *Ascophyllum nodosum* from the low intertidal at FE. *Codium fragile*, a large green alga, became and remains the dominant component of the FE community. It is expected that the present FE community will continue to be dominated by opportunistic, warm water-tolerant algae.

Millstone Point, the second-closest station to the discharges, has shown a decrease in *Fucus vesiculosus* coverage since 1981 when monitoring first began there. In addition, an increase in grazers, especially *Littorina littorea*, has been observed since the second cut was opened.

Recolonization studies, employing transects and exclusion cages, allowed isolation and identification of some factors that influence the structure of local rocky intertidal communities. Recolonization was influenced by time of year in which denuding occurred, and related to degree of exposure and intertidal height, e.g., rapid in the high intertidal of an exposed station and slow in the low intertidal of a sheltered station. Results from the exclusion cage studies have shown that rates and patterns of recolonization, both with and without the influence of grazers and predators, were unaffected by proximity to the discharge.

Growth and mortality studies of *Ascophyllum nodosum*, a perennial brown alga sensitive to water temperature change, were included in the rocky intertidal sampling program. Tip length analyses helped distinguish between a stressed population at Fox Island and populations at two reference stations. Since 1979, *Ascophyllum* plant loss averaged about 60% and tip loss about 80% overall; there was much variability in plant and tip mortality from year to year.

With the exception of the FE intertidal community, no significant changes to the benthic shore biota were observed that could be attributed to MNPS operation.

BENTHIC INFAUNA

Infaunal communities in the vicinity of MNPS were sampled to provide data needed to characterize subtidal and intertidal community abundance and species composition, identify spatial and temporal changes in these parameters and evaluate whether any observed changes were the result of construction and/or operation of MNPS.

Intertidal

Sediments at the three intertidal sampling stations were composed of medium sands that usually contained low amounts (< 3%) of silt/clay. During the baseline period, sediment grain size and the percentage of silt/clay were more consistent at stations exposed to constant wave-induced scour while at the seasonally protected Jordan Cove beach, these parameters exhibited considerable variability.

At all stations, infaunal communities were dominated by polychaetes and oligochaetes; these groups often accounted for over 75% of the total number of individuals collected annually. The polychaetes, *Scolecolepides viridis*, and *Polydora ligni*, and the oligochaete group were among the dominants at all stations. At Jordan Cove, oligochaetes accounted for over 60% of all individuals in 5 of the past 6 years and was the most abundant taxon in each of the last six sampling years. At Giants Neck and White Point, rhynchocoels, *Paraonis fulgens*, and *Haploscoloplos fragilis* were dominant, contributing over 10 percent of the total individuals over years.

Quarterly mean abundance (no./core) at intertidal stations ranged from 4 to 840 and mean species numbers from < 1 to 13. Overall, species abundances and species numbers were highest at Jordan Cove and of similar magnitude at Giants Neck and White Point. In general, intertidal abundance and species number were higher during warmer sampling periods (June and September) than during colder ones (March or December). Multiple regression analyses, which were based on natural abiotic variables were used to identify and remove naturally-induced temporal variations in abundance and number of species. After adjusting for this variation, no significant increasing or decreasing trends in abundance or species number were evident from 1980 to 1985.

Annual mean species diversity (H′) over the study period ranged from 1.2 to 2.3, and evenness (J) from 0.4 to 0.7, and on an annual basis, no consistent spatial or temporal shifts in H′ have occurred over the monitoring period. Diversity parameters reflected the spatial differences evident in the structure of intertidal communities throughout the monitoring program.

Data collected during the baseline period characterized the spatial relationships among sampling stations and investigated the extent and direction of temporal fluctuations in the abundance and numbers of species that have occurred from 1980 - 1985. During this period, no changes in intertidal communities could be attributed to operation of Millstone Units 1 and 2 or construction of Unit 3. Data presented here appear representative of natural, undisturbed intertidal communities and will be the basis for all future impact studies performed during 3-unit operating conditions.

Subtidal

Sediments at subtidal stations ranged from fine to coarse sands and contained up to 40% silt/clay. Sedimentary characteristics of all stations, except Jordan Cove, have exhibited some temporal shifts during the study. Values for grain size at Giants Neck have generally increased since 1982. At Intake, smaller grain size and more highly variable silt/clay values have been observed since June 1983, when construction activities (dredging and coffer dam removal) resulted in the deposition of fine material at this station. Silt/clay content at Effluent has also become more variable following construction activities in the area of the Unit 3 discharge cut.

Polychaetes and oligochaetes were the most abundant organisms at Effluent, Giants Neck and Jordan Cove while arthropods were frequently abundant at Intake. Polychaete abundances were generally highest in either September or June and although molluscs and arthropods contributed substantially to the subtidal communities no consistent seasonal trends were seen in these groups.

During 1986, higher numbers of arthropod species were recorded in September and December samples at Giants Neck and Jordan Cove than in previous years. At Intake, species numbers continued to increase in 1986 continuing a trend begun in 1985 after construction activities (in 1984) eliminated most of the species in this area. In contrast, the numbers of species and individuals of all the major taxa at Effluent were generally lower in 1986 than in the previous two years.

From March 1979 to March 1986, mean quarterly abundance of subtidal communities ranged from 9 to 633 (no./core) and mean species numbers (no./core) from 5 to 46. Highest quarterly abundances generally occurred in June or September at all stations. After removing naturally induced temporal variation, no significant increasing nor decreasing trends in annual mean abundance were evident at any subtidal station. Species numbers were generally highest in June or September at Effluent, Giants Neck and Jordan Cove; (December at Intake). At Effluent, the mean species number has increased significantly over the monitoring period.

Over all stations and years, oligochaetes (as a group) were the most consistently dominant taxon, accounting for 4 to 61% of the total individuals collected. Other taxa among the more consistently abundant forms were *Polycirrus eximius*, *Protodorvillea gaspeensis*, *Tharyx* spp., *Aricidea catherinae*, and *Tharyx acutus*. Of all communities, Intake was the most dissimilar; At this station, 29% of top ten dominants were species of amphipods. In addition, other dominant taxa at this station exhibited strong temporal fluctuations in abundance.

Mean annual species diversity (H′) ranged from 2.6 to 4.6 with lower values generally reflecting short-term pulses in species abundances. High H′ values reflected changes in the numbers of individuals or species, but not in the equality of the distribution of individuals among species.

The subtidal benthic infaunal program has provided data necessary to characterize communities inhabiting areas potentially impacted by power plant operations (Intake, Effluent, and Jordan Cove) and

an unimpacted area (Giants Neck). During the baseline period, temporal and spatial changes in species abundance and community composition were evident at all stations; however, those at Intake and Effluent appeared power plant related. Observed changes occurred prior to 1985, and were due to Unit 3 construction activities, including dredging, and not to operation of Millstone Units 1 and 2. The opening of the second discharge cut may have caused recent changes in sedimentary and community parameters observed at Effluent. The widespread nature of infaunal community changes at Giants Neck and Jordan Cove were limited mostly to rearrangements in the ranking of traditionally dominant species which suggest a response to naturally occurring events and are not related to operation or construction of the Millstone facility.

LOBSTER POPULATION DYNAMICS

The lobster population in the Millstone Point area was sampled using pots from 1976 to 1985. Lobsters > 55 mm carapace length were tagged and released to monitor growth and movement. Sex, presence of eggs, carapace length, missing claws and molt stage were also recorded. Tagged lobsters were released at the site of capture. These studies characterized population dynamics of the local lobster stock during two unit operation. In addition, studies of lobsters caught on the intake traveling screens, (impingement) and larvae drawn through the plants cooling water system, (entrainment) were conducted to assess these impacts of Millstone Units 1 and 2 on the local lobster population.

Annual total catch per unit effort (CPUE) ranged from 0.56 to 2.10 lobsters per pot from 1976 to 1985. The lower CPUE values corresponded to data collected with wood pots which allow small lobsters to escape between the 3-5 cm lath spacings. Wood and wire pots used during a 3 1/2 yr gear-comparison study provided the basis for a decision to use all wire pots in our lobster studies beginning in 1982. The CPUE of legal-sized lobsters (greater than or equal to 81 mm CL) was similar for wood and wire pots throughout the gear comparison study. With the exception of 1981, total CPUE was significantly higher for wire pots due to greater catches of sub-legal lobsters in wire pots. A special study conducted during 1982 to investigate the lower catch of wire pots used in 1981 indicated that trap efficiency was affected by the construction and placement of funnels used in pots. Other factors that contributed to the efficiency of lobster pots were the number of days between pothauls (soaktime) and the influence of competing species caught in pots. Mean monthly CPUE was adjusted accordingly using covariance analysis to account for these influences. Dredging activities in the vicinity of the intake

structures were responsible for lower catches at the Intake station during 1985, however, after the dredged area has stabilized lobsters will probably return to the area and catch rates at that station should increase in 1986. Based on the annual mean CPUE and inspection of the confidence intervals around the mean, the 1980 and 1981 CPUE's were the lowest annual values of all years from 1978 to 1985. Conversely, the 1982 CPUE was the highest annual value due to a strong prerecruit class in 1982 which sustained record landings throughout Long Island Sound during 1983 and 1984 when these lobsters molted to legal size.

Size frequency distributions indicated that wire pots caught significantly more small lobsters (< 75 mm) than did wood pots. Annual mean carapace lengths of lobsters caught in wire pots have been consistent (range 70.8-71.8). Since 1975, the overall sex ratio of males to females was close to 1:1 however, when three stations were compared, Twotree (1.5 km offshore) had consistently higher proportions of females, whereas Intake and Jordan Cove (< 0.5 km offshore) had slightly more males. Berried females comprised between 3.1 and 6.7% of all females caught from 1975 to 1985 and greater proportions were caught at Twotree. The size distribution of berried females collected and the abdomen width/carapace length relationship suggest that females first become sexually mature at about 50 mm carapace length and that all females are mature at 95 mm carapace length.

The number of molting lobsters observed in the weekly catch varied from year to year and over the sampling period. In general, molting peaked in June although in several years a fall molting peak was also observed. The average growth per molt of males (14.1%) and females (13.7%) was significantly different. The percentage of lobsters missing one or both claws (culls) ranged from 9.0 to 17.4%; more lobsters caught in wood pots experienced claw loss (14.4%) than lobsters caught in wire pots (12.7%). Since 1975, of the 57,359 lobsters caught; 47,259 were tagged and released, and 8,053 (17%) were recaptured in our sampling program. Commercial lobstermen caught an additional 13,394 (28.3%) of our tagged lobsters over the same period. About 95% of the lobsters were recaptured at the release station; movement between stations was minimal. Of the movement that occurred, most was inshore between Jordan Cove and Intake. Tagging studies indicate lobster movements are restricted to the local area since 91% of the commercial recaptures occurred within the study area; of those lobsters that were recaptured outside the study area, most moved to the east. Some lobsters traveled considerable distances, more than 100 km offshore, where they were caught on the edge of the continental shelf (Block and Hudson canyons).

Lobster larvae entrainment studies indicated that lobster larvae are susceptible to entrainment from mid-May through late-June. The occurence of lobster larvae in the cooling waters coincided with the peak abundance of berried females caught in our traps. More lobster larvae were collected in night samples (substantiated by two 24 h samplings in 1985). The fact that more larvae were collected at night when surface densities have been reported to be lowest may result from a combination of larval behavior and the intake structure curtain wall design. This design minimizes entrainment of lobster larvae during day when larvae are surface oriented. Survival of lobster larvae after passing through the plant's cooling water system was observed indicating that entrainment mortality is lower than the assumed 100%.

Since 1975, an estimated 11,359 lobsters were caught on the intake traveling screens. The number of lobsters impinged at Units 1 and 2 was highest in 1982, corresponding with the highest annual trap catch. A fish return system (sluiceway) was constructed at Unit 1 and began operating in December 1983 which improved survival and minimized damage to lobsters associated with the impingement process at MNPS.

There is no evidence to date that MNPS has significantly affected the local lobster population. Fluctuations in the annual abundance of lobsters throughout Long Island Sound and the variability in annual CPUE's of the local lobster population appear to be related to natural events.

EXPOSURE PANEL PROGRAM

Patterns of abundance and distribution of fouling and woodboring organisms on exposure panels at ambient water sites, determined during 2-unit operation, have been consistent from year to year, and were predictable based on seasonal water temperatures, and the life stages available for settlement. Wood-loss was caused primarily by *Teredo navalis*, and was highest during the May-Nov exposure period. Variation in identity and abundance of species that colonized the panel surfaces did not appear to affect recruitment or abundance of woodborers.

Effluent was also characterized by the absence of cold water species, e.g., *Laminaria saccharina*, and by the presence of a warm water shipworm, *Teredo bartschi*.

Teredo bartschi also maintained a reproductive population in panels located 100 m outside the quarry, exposed to the effluent produced by 2-unit operation. Exposure to elevated temperatures may be required for larval settlement. *Teredo navalis*, the native shipworm, was also found in the MNPS thermal effluent; there was a trend of increasing abundance with decreasing distance from the quarry.

Results from the Timber Study show that untreated wood is rapidly degraded in local waters, primarily by *Teredo navalis*; Red Oak is more resistent than Douglas Fir, but blocks with a minimum dimension of 6.4 cm have a survival time on the order of 2-3 yrs.

Chemical treatment can deter woodborer attack, but if timbers are cut after treatment, unprotected surfaces are exposed, and are susceptible to woodborers. This susceptibility is less for creosote treated wood than for CCA treated wood, as creosote is a better penetrant.

Characteristics of fouling and woodboring communities in the vicinity of MNPS have been established during 2-unit operation. Comparisons of community and population parameters determined after Unit 3 is operational will permit assessment of the potential added impact.

FISH ECOLOGY

The construction and operation of MNPS could affect fish assemblages in several ways. Larger fish may be removed from the population by impingement on the intake screens; eggs, larvae and small fish may be removed during entrainment through the cooling water system; and spatial distribution of local fish populations may change in response to the cooling water effluent.

Several programs were established to provide baseline data for assessing impacts of MNPS on fish assemblages. These include studies of planktonic, demersal, pelagic and shore-zone fish assemblages, and estimates of the number of fish impinged and entrained. Plankton studies conducted since 1973 included collections of fish larvae at various stations, and entrainment mortality and thermal tolerance research on selected larval fish taxa. The trawl sampling program was established in 1973 to monitor spatial

Several programs were established to provide baseline data for assessing impacts of MNPS on fish assemblages. These include studies of planktonic, demersal, pelagic and shore-zone fish assemblages, and estimates of the number of fish impinged and entrained. Plankton studies conducted since 1973 included collections of fish larvae at various stations, and entrainment mortality and thermal tolerance research on selected larval fish taxa. The trawl sampling program was established in 1973 to monitor spatial and temporal fluctuations of demersal fish. The gill net program, started in 1971 to provide qualitative estimates of local pelagic fish assemblages, was dropped at the end of 1982 because catches were generally low and none of the species collected were adversely affected by MNPS. The seine sampling program was established in 1969 to monitor shore-zone fish. Impingement monitoring began at MNPS Unit 1 in 1972 and at Unit 2 in September 1975 and was supplemented by several fish diversion and survival studies. These programs, which provide the data necessary for assessing the effects of two-unit operation also provide the baseline for three-unit impact assessment.

Over 100 taxa of fish have been collected in the various Fish Ecology monitoring programs at MNPS from January 1976 through December 1985. Composition of the fish assemblages studied during that period remained relatively stable and were typical of those reported for LIS by other researchers. Eight taxa were selected for detailed analyses based on their susceptibility to impact from impingement and entrainment: anchovies, sand lance, sticklebacks, silversides, tomcod, grubby, cunner and tautog.

The abundance of these taxa varied both seasonally and annually in all programs and to separate fluctuations representing natural variability from those resulting from the construction and operation of MNPS, a time-series approach was developed and applied to the monitoring data. This approach, which combined several statistical techniques (harmonic regression, analysis of variance and time-series analysis) to summarize catch fluctuations in the long-term data series, provided confidence intervals that were narrower than those associated with annual or monthly means or medians and was, therefore, more sensitive to unusual abundance fluctuations.

The abundances of potentially impacted taxa remained relatively stable throughout the 10-year period, except for larval and juvenile sand lance, and larval anchovy, cunner and tautog. Except for larval sand lance, these abundance changes were short-term. Larval sand lance abundance decreased during 1982 and has remained low since then. A mass impingement (390,000) of sand lance juveniles occurred in a one week period in July 1984 but was an uncommon event and is not expected to recur.

In 1984, there was a marked decrease in the abundance of larval anchovy, cunner and tautog but this decrease was also observed in other parts of LIS. The reasons for this decline were not known but during this time ctenophores, a plankton predator, were abnormally abundant.

Impacts from the construction and operation of MNPS were assessed using representative collections of fish assemblages during the operation of Units 1 and 2. Except for larval sand lance, there was no indication that catches of the most abundant fish taxa were consistantly below historic levels and for sand lance, the observed fluctuation occurred along the entire Atlantic coast. Thus the operation of two nuclear power plants at MNPS has not adversely affected fish abundance, distribution or species composition in the Millstone area of LIS.

WINTER FLOUNDER STUDIES

The life history and population dynamics of the winter flounder (*Pseudopleuronectes americanus*) have been studied intensively since 1973, due to its importance to the sport and commercial fisheries of Connecticut and potential for impact. Because winter flounder stocks are localized, most work has concentrated on the population spawning in the Niantic River to determine if MNPS impacts of impingement and entrainment have caused or would cause changes in abundance beyond those expected from natural variation.

Annual estimates of the Niantic River spawning population have been made since 1976. An abundance index based on the stochastic model of Jolly for open populations showed that numbers were relatively stable from 1976 through 1980, increased to a peak in 1982, and subsequently declined to an 11-yr low in 1986. Abundance determined by trawl CPUE generally paralleled the Jolly index through 1982. The decline in CPUE was greater through 1985 and less in 1986 than for the corresponding Jolly estimates. The influence of potential biases on both estimators were examined. A third measure of abundance was provided from trawl monitoring program data using time-based harmonic regression models. However, models from most stations were unsatisfactory due to insufficient data or a lack of a repetitive pattern of abundance. High variability in catch, relatively low effort, and the mixture of stocks found at most stations at certain times of the year make these trawl data difficult to interpret and of limited use in assessing MNPS impact on the winter flounder. Throughout southern New

England, winter flounder abundance has declined in recent years because of natural fluctuations and also most likely because of recent increases in commercial fishing.

As reported for other populations, the average sex ratio for Niantic River winter flounder was 1.44 in favor of females. The length of 50% maturation of females was 26.8 cm, equivalent to age 3 or 4. Most spawning in the Niantic River was completed by early April with annual variations apparently related to water temperature. Egg production was a function of female size and the length-fecundity relationship was similar to those reported for other populations. Egg production peaked in 1982 and has since decreased about 80%.

Scales were successfully used to age winter flounder. Mean lengths of age 3 and older females were significantly larger than those of males. Growth was relatively rapid in early years, but older age groups overlapped considerably in size. Growth of the Niantic River fish was less than other populations in the region through age 2, but equaled or exceeded their means at age 3 and older. The von Bertalanffy model was used to calculate population growth parameters using 1983 length-at-age data. L_∞ was determined as 423 and 381 mm and K as 0.42 and 0.44 for females and males, respectively. The mean annual survival rate of age 3 and older adults was determined as 0.486 using a catch curve with samples combined from successive years to reduce bias. As found elsewhere, the winter flounder preyed upon a variety of benthic organisms and algae. Food items varied by location and reflected bottom type and different benthic communities.

The overall rate of return of Petersen disc-tagged winter flounder was 25%. About twice as many were taken by the sport than the commercial fishery, although less cooperation was probably received from the latter. Most (70%) of the returns were from local waters and three times as many of the longer-distance recaptures were made in waters to the east than to the west.

Direct tissue isoelectric focusing techniques were used to differentiate stocks of winter flounder. Good separation was achieved using fish from major estuaries in Connecticut and Rhode Island at least 8 km apart. A second study using fish from areas closer to MNPS showed more homogeneity, with significant intermixing occurring throughout much of the year. The technique could not be used to separate immature specimens.

Several special studies and analyses were conducted to identify possible sampling biases in the larval winter flounder data base. The results of these studies included reduced larval net extrusion with 202-μm mesh nets compared to 333- and 505-μm nets, increased sample density of larger larvae in night collections, and changes in sample densities in relation to tidal stage at a station in the lower portion of the Niantic River. Due to the identified sample biases, much of the offshore data collected prior to 1980 could not be used to examine the life history of larval winter flounder.

Based on the abundance and distribution of smaller larvae, spawning primarily occurred in the Niantic River. Larvae were gradually flushed into Niantic Bay, where larger larvae dominated. The spatial distribution of larvae within the Niantic River varied from year to year, but generally smaller larvae were more prevalent in the upper portion of the river and larger larvae in the lower. The lion's mane jellyfish was identified as an important predator of larval winter flounder.

Eight tidal export-import studies were conducted at the mouth of the Niantic River during 1983-85. The results showed a net export of 4 mm and smaller winter flounder larvae and a net import of 5 mm and larger larvae. Larvae with developed fin rays migrated vertically in response to tidal currents to reenter the Niantic River and those within the river demonstrated a similar behavior as a retention mechanism.

Examination of otoliths from field-collected and laboratory-reared winter flounder larvae indicated that daily increments were not visible. Based on the length-frequency distribution, most larval mortality occurred at the time of first feeding (3-4 mm). Transition to the demersal juvenile stage occurred at about 6-7 mm.

Abundance of post-larval young-of-the-year peaked in mid-June and stabilized by late July. Young were most numerous in the lower river during 1983, with similar densities found during 1984 and 1985. Growth of young in the lower river was significantly greater than at stations farther upriver after mid-June. Weekly mean lengths in 1983 were about 6 to 8 mm larger than in 1984 or 1985. Monthly survival estimates of young ranged from 0.552 to 0.569 in the lower river and 0.661 at a station in mid-river.

Peak abundance of age 1 juvenile winter flounder taken in the Niantic River during the adult surveys occurred in 1981, with second and third highest CPUE in following years. An 11-yr low was

found in 1986. However, in recent years juveniles have been found in more areas throughout the river and in Niantic Bay during the time of the surveys. This variation in distribution makes the estimation of juvenile abundance less certain than that of adults.

About two-thirds of the total number of winter flounder impinged on the traveling screens of MNPS were taken in winter. Before 1984, annual estimates usually ranged from 4 to 10 thousand with winter storms accounting for large proportions of most annual totals. Sex ratios and reproductive condition of impinged fish differed from fish taken in the river. The predominance of males and of gravid females in the collections indicated that at times impingement was related to behavior of winter flounder. A fish return sluiceway was installed at MNPS Unit 1 in December of 1983 and studies showed that survival of returned winter flounder would be considerable (ca. 80-90%). This greatly reduces the impact of impingement on the winter flounder.

Entrainment sampling has been conducted since 1976. A majority (> 60%) of the winter flounder larvae entrained were 5 mm and larger. The greatest entrainment densities occurred from mid-April through May. Based on the median annual entrainment density, three years were low (1977-79), four years were high (1976, 1980-83), and the remaining years were intermediate. Annual entrainment was related to total egg production in the Niantic River and the length of time a larva was susceptible to entrainment was related to water temperature. The effects of entrainment on larval winter flounder were examined in the laboratory and field. Larvae 5 mm and larger were able to survive a ΔT of 13 °C for up to 9 h. The estimated critical thermal maximum was approximately 24 °C. A mortality study showed that about 80% of the Stage 4 larvae would have survived entrainment.

Impact assessment was addressed using a deterministic model developed by the University of Rhode Island. The model, subdivided into hydrodynamic, concentration, and population submodels, predicted a 5 to 6% decrease in the Niantic River population after 35 yr of MNPS operation. Based on the initial assumptions, the model results were probably conservative. A new stochastic population dynamics model, which takes into account the natural variability in the recruitment process, is currently under development at NUEL and will provide a more realistic estimate of potential losses.

To date, there is no evidence that MNPS has significantly affected the local winter flounder population. Variability in annual abundance appears to be related to natural events and has been noted

throughout the region. Future work at NUEL will focus on early life history stages and will include estimates of larval mortality, which are critical to the assessment of MNPS impact.

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Contents

INTRODUCTION 1
BIBLIOGRAPHY 9

INTRODUCTION

The Millstone Nuclear Power Station (MNPS) is located on the north shore of Long Island Sound (LIS) in Waterford, Connecticut. The station consists of three units located on a peninsula bounded by Jordan Cove on the east and by Niantic Bay on the west (Fig. 1). Millstone Unit 1, which commenced commercial operation November 29, 1970 is a 652-MWe boiling water reactor (BWR). Unit 2, an 870-MWe pressurized water reactor (PWR), began operating October 17, 1975. Construction of Unit 3, a 1,150-MWe PWR, began in August 1974; commercial operation began April 23, 1986.

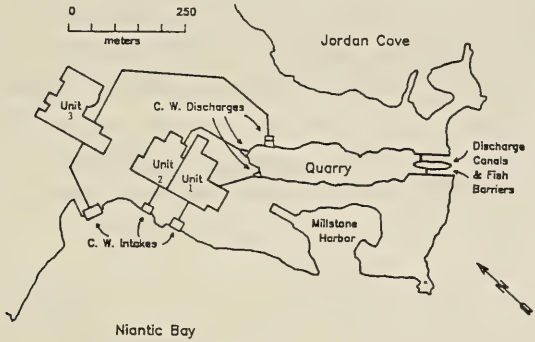

Figure 1. Site plan of the Millstone Nuclear Power Station.

All three units use once-through condenser cooling water systems. The rated circulating flows for Units 1, 2 and 3 are 935, 1,220 and 2,000 cfs, respectively. Cooling water is drawn from depths greater than four feet below mean sea level by separate shoreline intakes located on Niantic Bay. The intake structures, typical of shoreline installations, have course bar racks and traveling screens. The cooling water, heated to 17 °C above ambient, flows from discharge structures through an abandoned granite quarry and exits into LIS through two channels equipped with fish barriers. Cooling water flow rates from 1976 through 1985 are presented in Figure 2 for Units 1 and 2. Flow rates for Units 1 and 2 prior to 1976 are provided in Appendices on file at Northeast Utilities Environmental Laboratory (NUEL). Ambient water temperatures and ΔT for the same period are presented in Figure 3.

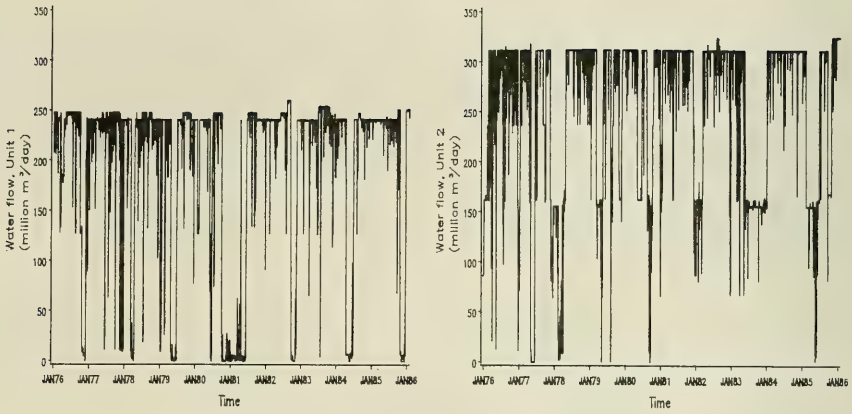

Figure 2. Cooling water flow rates for Units 1 and 2 from 1976 to 1985.

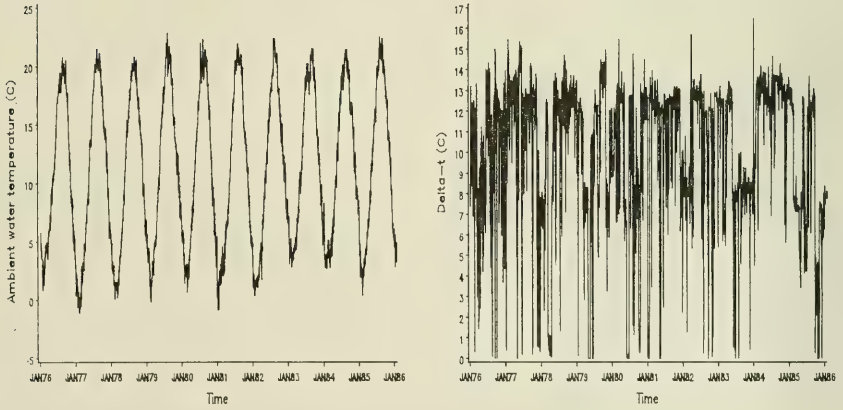

Figure 3. Ambient water temperature and ΔT during two unit operations (1976-85).

The potential impact of MNPS on LIS biota has been the focus of study since 1968. The early biological investigations included exposure panel monitoring of woodboring and fouling communities, and surveys of the intertidal sand, rocky shore and shore-zone fish communities. The program scope increased considerably between 1970 and 1973 with the addition of heavy metal analyses of seawater and mollusc tissue, studies of impingement, pelagic and demersal fishes, plankton, subtidal benthos, and lobster and Niantic River winter flounder population studies (Battelle - W.F. Clapp Laboratories 1975; NUSCo 1975). A bibliography of all Millstone-related studies is provided at the end of this introduction; results of these studies are on file at NUEL.

Studies of entrained plankton began in 1970 when Unit 1 became operational (Carpenter 1975); studies at Unit 2 began in 1975. To date, the routine monitoring and special investigations have covered nearly all components of the plankton, including ichthyoplankton, phytoplankton, and zooplankton. Effects of chlorination and increased temperature on entrained phytoplankton were addressed and latent mortality of zooplankton after condenser passage was determined (Carpenter et al. 1972b, 1974b) Later, emphasis was placed on entrained ichthyoplankton and the relative impact of entrainment on fish populations in surrounding waters (NUSCo 1976b, 1984a).

Impingement monitoring began at Unit 1 in 1971 and at Unit 2 in 1975. The scope has varied from a complete census of all impinged organisms (1972-1976) to the current program of counting those organisms impinged during a 24 h period on several days per week. Special studies have evaluated the effectiveness of several fish deterrent systems at the intakes, including acoustics, underwater lighting and a surface and bottom barrier (NUSCo 1976b, 1980a, 1981a). In December 1983, a fish return system (sluiceway) began operating at Unit 1 reducing impingement related impacts on fish and shellfish populations (NUSCo 1981b, 1986b).

The potential effect of three-unit operation on selected species was also considered. Mathematical population dynamics models were developed for the Niantic River winter flounder population (Hess et al. 1975; Saila 1976) and for the regional menhaden population (NUSCo 1976b, 1983c). These models estimated the effect of predicted entrainment and impingement losses to populations over the life of the power station.

Several hydrographic and hydrothermal studies have been conducted since 1965 (NUSCo 1983c). These studies were of two types. Some were designed to predict the shape and extent of the plume under 1-, 2-, and 3-unit operations. Others were field studies designed to determine the general hydrographic nature of the waters surrounding MNPS or to verify the shape and extent of the thermal plumes (NUSCo 1979d). In addition, a tidal circulation model was developed that predicted current patterns and thermal distributions and simulated dispersal and entrainment of Niantic River winter flounder larvae (Hess et al. 1975; Saila 1976).

As a result of these studies, the hydrographic and ecological characteristics of surrounding waters are described. Studies have been intensified and modified to provide the most representative data with respect to the changing concerns and state-of-the-art techniques. The present report provides results of studies conducted during two-unit operations and provides a basis for evaluating any long-term impacts associated with three-unit operations at Millstone Point. The report also satisfies certain license and permit conditions stipulated by the Connecticut Department of Environmental Protection and the Connecticut Power Facility Evaluation Council.

All ecological and hydrographic studies through 1976 were conducted by consulting laboratories, most notably Battelle - W.F. Clapp Laboratories, Woods Hole Oceanographic Institute and Normandeau Associates. In 1977, Northeast Utilities Service Company (NUSCo) began a phased, in-house takeover beginning with the entrainment and impingement programs. Some benthic and lobster program responsibilities were added in 1978. As of January 1980, all studies (excluding heavy metals) were being conducted and reported by NUSCo scientists based at NUEL. Critical scientific review is provided by a four-member, Ecological Advisory Committee (see acknowledgements) that has provided continuing support since 1968.

Data from the MNPS monitoring programs, the results of which are presented in this summary report, are stored on discs at NUEL and at Environmental Programs Dept. in Rocky Hill, CT. Appendices to this report are also archived at NUEL.

In addition to the monitoring programs described in this report, several studies were conducted in earlier years but were later discontinued (i.e., phytoplankton, zooplankton, heavy metals). A brief summary of these studies' results follows; more detailed results can be found on file at NUEL.

PHYTOPLANKTON - Phytoplankton, as primary producers, were considered to be an important component of the marine ecosystem to study in early power plant impact assessments. Thus, various studies were completed at MNPS from 1970 to 1982 to monitor and assess the impact of entrainment on phytoplankton community composition and abundance. The phytoplankton community in the vicinity of Millstone was found to be similar to that of Long Island Sound and to that near Cape Cod (NUSCo 1983b). Carpenter (1975) found that the higher discharge temperature depressed productivity only during warmer periods at Millstone. Chlorination had the greatest impact on phytoplankton, but during chlorination the predicted 5-10% decrease in productivity and biomass in the effluent mixing zone at Millstone could not be detected (Carpenter et al. 1974a). This result was not unexpected because NUSCo (1979a) estimated that, assuming total mortality of entrained phytoplankton and using published growth rates, the phytoplankton populations in the vicinity of MNPS could recover to preentrainment levels in 1 to 9 h. Because it became apparent that power plant operation had a negligible impact on the phytoplankton community, the phytoplankton program was discontinued in 1982 (NUSCo 1983a).

ZOOPLANKTON - Zooplankton densities in the Millstone area and potential entrainment losses were also examined as part of the long term monitoring program because of the importance of zooplankton to the marine ecosystem. Zooplankton studies were conducted from 1970 through 1983 and included, estimates of zooplankton densities at several nearshore stations in and around Niantic Bay, estimates of entrained zooplankton, and studies of entrainment-related zooplankton mortalities (NUSCo 1983a). Carpenter et al. (1974b) found that entrainment resulted in zooplankton mortality but significant changes in the zooplankton community near MNPS were not observed. Only a small percentage of the community was directly influenced by power plant operation (4% of the average volume of the Niantic Bay tidal exchange (NUSCo 1976b) and the densities and species composition were typical of greater LIS. Because it was unlikely that detectable changes in zooplankton species composition or abundance would occur during Unit 3 operation, the zooplankton monitoring programs were discontinued in 1983 (NUSCo 1983c).

OSPREY - The osprey (*Pandion haliaetus*) is a pisciverous raptor found in many estuarine areas along the east coast of North America. The area around MNPS is an established breeding ground for osprey; both Jordan Cove and the Niantic River provide abundant food. Because the population in the northeast declined during the 1950's and 1960's as a result of egg shell thinning from the ingestion of DDT, federal officials placed the osprey on the list of threatened species. To assist the recovery of osprey populations after the ban on DDT, NUSCo erected nesting platforms on MNPS property between 1967 and 1985

(NUSCo 1985a). Since then, 61 young have been produced from these nests; over 400 were produced in Connecticut (NUSCo 1985a). NUSCo continues to monitor the recovery of these magnificent birds.

HEAVY METALS - Concentrations of heavy metals in seawater and shellfish tissue samples were monitored five times per year from 1971 through 1983. Seawater and shellfish tissue samples were examined for concentrations of copper, zinc, iron, chromium, and lead at areas adjacent to and distant from the MNPS to assess possible heavy metal additions associated with seawater passage through the plant's cooling water systems.

Results indicated enhanced levels of copper, nickel, and zinc in cooling water samples relative to levels in seawater samples collected outside the immediate mixing zone of the MNPS plume (NUSCo 1983b). However metal levels return to ambient at rates similar to the return of water temperature to ambient. Of all shellfish tissue samples analyzed, only those from oysters inhabiting the quarry had elevated metal concentrations. In general a decline in concentrations of soluble and insoluble phases of heavy metals was observed and was related to improved analytical techniques rather than actual decreased levels in concentrations (Waslenchuk 1980, 1981, 1982, 1983). For example, concentrations in sea water reported in 1982 were significantly lower than reported in 1981, under similar plant operating conditions (NUSCo 1983b). Since concentrations of metals found in shellfish outside of the effluent quarry were comparable during one and two unit operations, and since Unit 3 has a titanium condenser that resists corrosion, the heavy metals monitoring program was discontinued in 1984.

AQUACULTURE - In 1976, a study was initiated to assess the feasibility of utilizing the thermal effluent in the culture of selected species of shellfish. The bay scallop (*Argopectin irradians*) was chosen as the study species because it could be easily cultured in the laboratory and was a desired sport and commercial species locally. Bay scallops were successfully reared from egg to juvenile stages utilizing effluent waters at Millstone Point (MRI 1980). Discharge waters 11-12 °C above ambient temperatures favored the conditioning of brood stocks for early spring and late fall spawnings. This, coupled with increased growth rates of juveniles cultured in the effluent during winter months, potentially provided a method of seeding coastal waters with juvenile shellfish. Initial experiments to seed hatchery reared juveniles into nearby Jordan Cove indicated that predation of young scallops was high and would be the limiting factor in the success of any put-and-take type of fishery.

BIOASSAY - In 1981, continuous effluent toxicity testing of the discharge from MNPS began using populations of sheepshead minnow (*Cyprinodon variegatus*) and mysid shrimp (*Mysidopsis bahia*) subjected to undiluted effluent in quarry waters and to Jordan Cove water (as a control), both adjusted to constant temperature of about 20 °C. Both organisms have been extensively used in marine toxicity testing and are among EPA recommended test organisms. Because the effluent is primarily condenser cooling water, it has a low potential for toxicity. Potential toxicants that are present in the MNPS effluent include chlorine, heavy metals and hydrazine.

Sheepshead minnow embryo-larval tests were used to assess acute toxicity of the discharge because the early life history stages of fish are most sensitive to toxicants. Sheepshead minnow life cycle tests were also conducted to assess chronic toxicity by examining egg viability, larval mortality and growth. Parameters examined during *Mysidopsis bahia* toxicity testing were population growth rates, and reproductive potential. Comparisons were made between animals maintained in undiluted MNPS effluent and Jordan Cove water.

Results of these studies to date indicate no chronic toxicity related to the MNPS effluent. No differences were found between control and effluent treatments for egg viability, and development and growth of larvae (NUSCo 1983f, 1986c). Effluent toxicity testing will continue during three unit operation.

SHELLFISH SURVEY - During 1984 and 1985, the abundance and distribution of edible bivalves, hard clams (*Mercenaria mercenaria*), soft-shell clams (*Mya arenaria*), eastern oysters (*Crassostrea virginica*), and bay scallops (*Argopectin irradians*) was documented. These surveys were conducted in three steps at intertidal and subtidal areas around MNPS to establish a data base for these commercially valuable bivalves prior to three unit operations and to verify the distribution of bivalves in Jordan Cove previously reported in the State of Connecticut Shellfish Concentration Area Maps (NUSCo in prep.).

Survey results indicated that hard clams were the most abundant and valuable shellfish. In addition to the target species assessed in these studies, the razor clam (*Ensis directus*) was collected and was the second most abundant species, however, it is not a commercially or recreationally harvested species in the Millstone Point area. Soft-shell clams occurred at lower densities and supported a marginal recreational fishery in Jordan Cove. Only a single eastern oyster was found during the survey; no bay scallops were found in the study area although an important scallop fishery occurs in the Niantic River (about 2 km from MNPS).

EELGRASS - *Zostera marina* is an important component of estuarine systems because it stabilizes sediments and provides habitat for many marine organisms. In the early 1970s two studies were sponsored by NUSCo and conducted by the University of Connecticut to document eelgrass distribution in the MNPS area (Klotz and Knight 1973; Knight and Lawton 1974). In 1985, NUSCo conducted a more extensive survey which included mapping the extent of eelgrass in Jordan Cove, estimating standing stock and monitoring plant density, length and reproductive status. These studies were conducted to compare results of earlier eelgrass distribution and provide a quantitative data base prior to three-unit operations to assess possible impacts on eelgrass after Unit 3 start-up. Initial results of the NUSCo studies show average eelgrass densities ranged from 18 to 39 plants/m^2 and average blade length ranged from 144-681 mm. Standing stock estimates at Jordan Cove averaged from 39.4 to 297.7 g/m^2 and were highest in July. Effects of the thermal plume on the Jordan Cove eelgrass stock have not been evident during two unit operating conditions.

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Contents

ROCKY INTERTIDAL STUDIES 1
INTRODUCTION 1
MATERIALS AND METHODS 2
RESULTS AND DISCUSSION 8
Temperature Data 8
Qualitative Studies 12
Quarry Study 20
Undisturbed Transects 22
Recolonization Studies 38
Transects 38
Exclusion cages 45
Ascophyllum nodosum Studies 50
Growth 50
Mortality 51
SUMMARY 59
CONCLUSION 60
REFERENCES CITED 61

ROCKY INTERTIDAL STUDIES

INTRODUCTION

The intertidal zone and near-shore waters are among the most productive regions of the world (Mann 1973). Intertidal algae provide food directly and indirectly to snails, crabs, and other benthic invertebrates, as well as to fish, shore birds, and man (Paine 1980; Edwards et al. 1982; Menge 1982). Some algae, *Ascophyllum nodosum* for example, release a large portion of their annual biomass as detritus and dissolved nutrients (Josselyn and Mathieson 1978). Other algae, primarily annual species, are consumed directly. Large perennial algae also contribute to the intertidal community's physical structure by providing shade and protection to much of the shore biota, and attachment space for epiphytes (Lewis 1964; Stephenson and Stephenson 1972; Menge 1975; Lobban et al. 1985).

Gradients of many parameters affecting shore populations result in universal patterns of zonation (Chapman 1946; Lewis 1964; Zaneveld 1969; Stephenson and Stephenson 1972). Some of these parameters can be quantified and characterized over time, and some can be experimentally manipulated in an effort to determine causal relationships (Connell 1961; Paine 1966; Dayton 1975; Menge 1975).

Rocky intertidal communities have certain attributes that make them ideal subjects for ecological assessment (MYAPCo 1978; Wilce et al. 1978; LILCo 1983; PSNH 1985). The stability of rocky shores permits establishment of permanently marked sampling areas. A discrete segment of a community, in many cases the same individual plants and animals, can be studied by successive observations. Some shore species are long-lived, capable of integrating effects of environmental conditions over their life spans. The presence of ephemeral species, which respond quickly to environmental conditions, reflects environmental change or instability. Sessile and slow-moving species are continuously exposed to potential impacts; others are motile, whose occurrence and abundance at a locality indicates the suitability of the environment at a given time. Many intertidal species show precise seasonal patterns of occurrence, abundance, and reproductive status. These seasonal patterns allow a multitude of spatial and temporal biological comparisons (Vadas et al. 1978; Schneider 1981).

The intertidal region in the vicinity of Millstone Point is exposed to a potential thermal impact from MNPS. Response to an impact may be obvious or subtle, and may occur at a community, population, or species level. The Rocky Shore monitoring program was designed and implemented with the following objectives:

1. to identify the attached plant and animal species found on nearby rocky shores,

2. to identify and quantify temporal and spatial patterns of occurrence and abundance of benthic species on these shores, and

3. to identify the physical and biological factors that induce variability in these intertidal areas.

To achieve these objectives, the rocky intertidal studies include qualitative algal collections, abundance measurements of intertidal organisms (percent of substratum covered), measurement of rates and patterns of recolonization following small-scale perturbation, experimental exclusion of grazers and predators for selected areas, and growth studies of *Ascophyllum nodosum*. These studies permit determination of potential biological perturbation from operation of Units 1 and 2 and construction of Unit 3. These studies also provide base-line data that will permit prediction and assessment of additional impact, if any, from operation of Unit 3.

The purpose of this report is to provide a summary of results from all rocky shore studies performed during 2-unit operation. Space limitations required considerable condensation of information; complete data are included in Appendix RS Ia-d.

MATERIALS AND METHODS

The rocky intertidal sampling program at MNPS began in May 1968 (Fig. 1; NUSCo 1982); the initial surveys were primarily qualitative in nature. In August 1978, the sampling program was evaluated, and extensive modifications were proposed (Appendix RS II, Proposed Changes for Intertidal Rock Sampling at Millstone Point). Modified sampling procedures were instituted in February 1979, incorporating non-destructive sampling, emphasizing more frequent qualitative and quantitative collections. This report

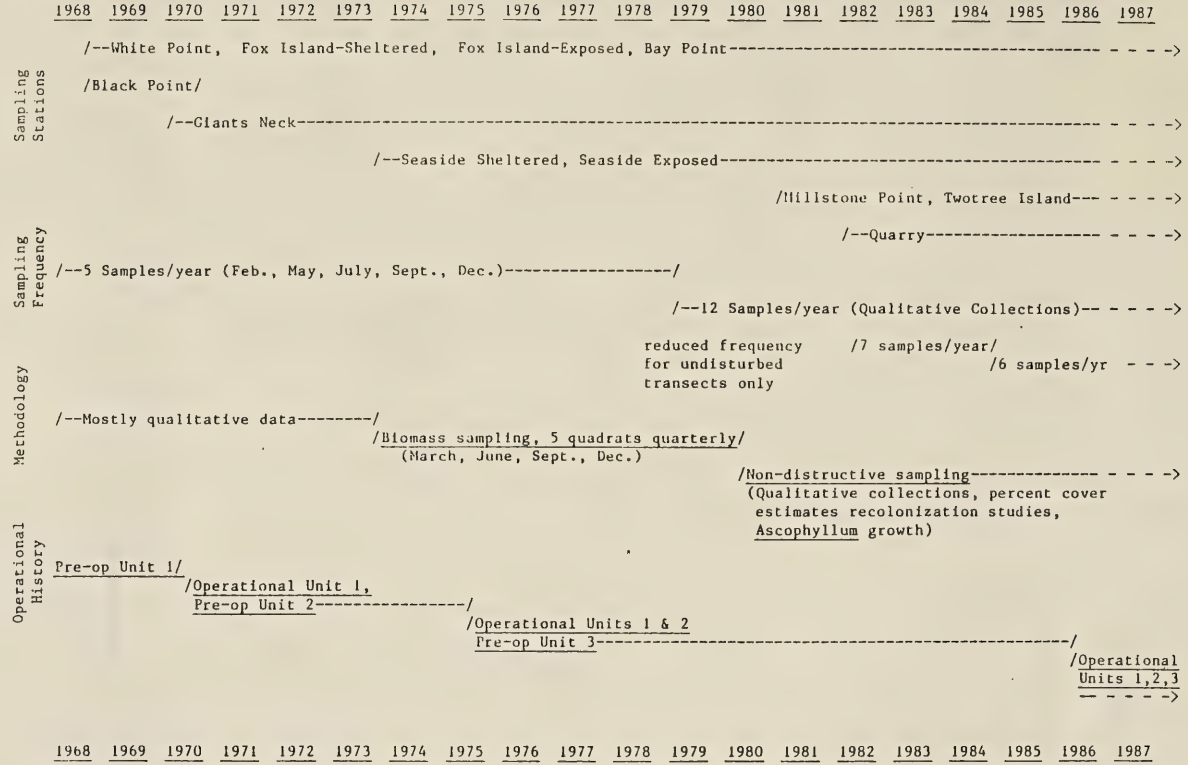
1968 1969 1970 1971 1972 1973 1974 1975 1976 1977 1978 1979 1980 1981 1982 1983 1984 1985 1986 1987
Sampling Stations
/--White Point, Fox Island-Sheltered, Fox Island-Exposed, Bay Point----------
/Black Point/
/--Giants Neck----------
/--Seaside Sheltered, Seaside Exposed----------
/Millstone Point, Twotree Island---
/--Quarry----------
Sampling Frequency
/--5 Samples/year (Feb., May, July, Sept., Dec.)----------/
/--12 Samples/year (Qualitative Collections)--
reduced frequency for undisturbed transects only
/7 samples/year/
/6 samples/yr
Methodology
/--Mostly qualitative data--------/
/Biomass sampling, 5 quadrats quarterly/
(March, June, Sept., Dec.)
/Non-distructive sampling----------
(Qualitative collections, percent cover estimates recolonization studies, Ascophyllum growth)
Operational History
Pre-op Unit 1/
/Operational Unit 1, Pre-op Unit 2----------/
/Operational Units 1 & 2 Pre-op Unit 3----------/
/Operational Units 1,2,3
1968 1969 1970 1971 1972 1973 1974 1975 1976 1977 1978 1979 1980 1981 1982 1983 1984 1985 1986 1987

Figure 1. Historical synopsis of Rocky Intertidal Survey.

summarizes data collected from March 1979 through February 1986, and characterizes local rocky intertidal communities under 2-unit operating conditions.

Qualitative Collections

The benthic algal flora at nine rocky intertidal stations (Fig. 2) was monitored qualitatively on a monthly basis. These stations are, in order of most exposed to least exposed: Bay Point (BP), Fox Island-Exposed (FE), Millstone Point (MP), Twotree Island (TT), White Point (WP), Seaside Exposed (SE), Seaside Sheltered (SS), Giants Neck (GN), and Fox Island-Sheltered (FS). Sampling at MP and TT began in October 1981.

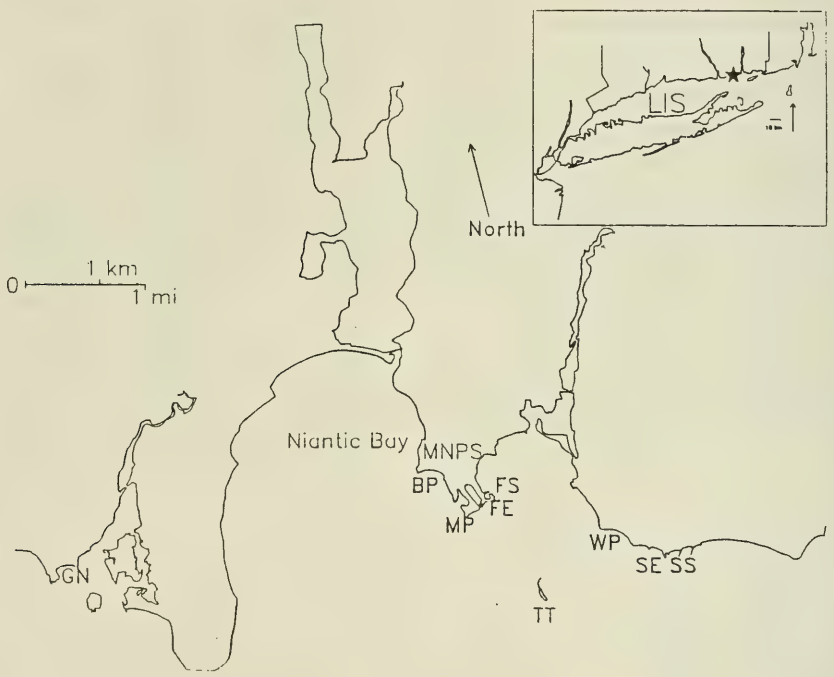

Figure 2. Location of rocky intertidal sampling sites. GN = Giants Neck, BP = Bay Point, MP = Millstone Point, FE = Fox Island-Exposed, FS = Fox Island-Sheltered, TT = Twotree Island, WP = White Point, SE = Seaside Exposed, SS = Seaside Sheltered.

Qualitative collections were made over an area sufficiently wide to characterize the flora at each site. Samples were identified fresh, or after short-term freezing. Voucher specimens were preserved using various methods, depending on the material: in 4% formalin/seawater, as dried herbarium mounts, or on microscope slides.

The Millstone quarry study was initiated in 1982. One site at the original quarry cut (site 1, Fig. 3) and three sites within the quarry (sites 2-4) were sampled monthly to permit examination of species composition under greater than ambient water temperatures. The quarry samples were processed as described for the qualitative collections from the rocky intertidal stations.

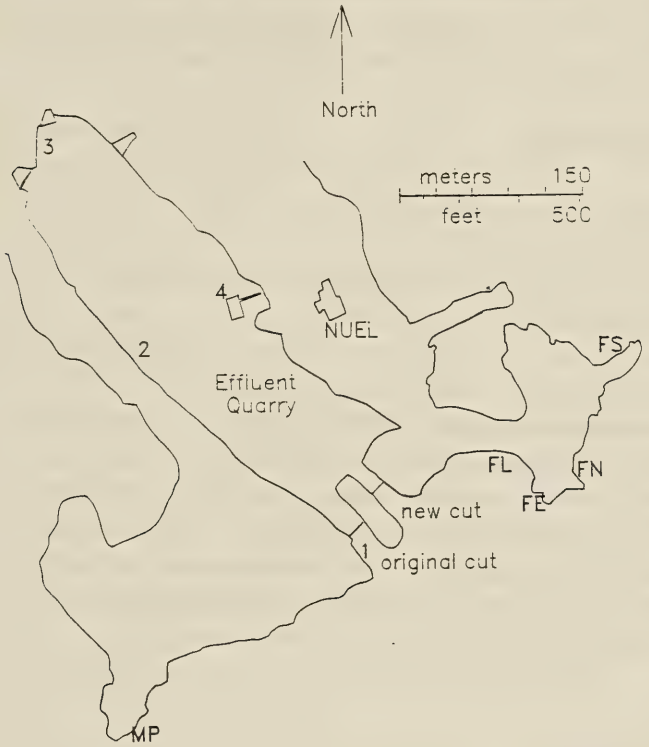

Figure 3. Detail map of MNPS vicinity. FL = original experimental *Ascophyllum* site (1979-1984), FN = new experimental *Ascophyllum* site (1985-present), 1-4 = effluent quarry collection sites.

Quantitative Studies

At each qualitative collection station except Twotree Island (owing to insufficient exposed bedrock), five permanent transects were established perpendicular to the water-line, one-half meter wide and extending from Mean High Water to Mean Low Water levels. Each transect, composed of 0.5 x 0.5 m quadrats, was non-destructively sampled monthly from March 1979 to February 1981, seven times per year from March 1981 to April 1983, and six times per year, in odd numbered months since May 1983 (see Appendix RS III, Environmental Tech. Spec. Change Request for Intertidal Rocky Shore Survey). The percentage of substratum cover of all organisms and remaining free space in each quadrat was subjectively determined and recorded. Understory organisms, or species that were partially or totally obscured by the canopy layer, were assigned a percentage that reflected their true abundance.

Recolonization Studies

Transects

Rates and patterns of recolonization following substratum denudation were determined in two series of recolonization transect experiments that were conducted at four stations (Fox Island-Exposed, Fox Island-Sheltered, White Point, Giants Neck). Sample design included two pairs of stations with similar degrees of exposure (Fig. 2): exposed at FE and WP, and sheltered at GN and FS. The Fox Island stations, because of their proximity to the MNPS discharge, were considered potentially impacted, while White Point and Giants Neck were identified as reference stations. In April 1979, three vertical transects were established at each station. Each transect was scraped free of attached algae and invertebrates and burned with a liquid petroleum gas torch. All recolonization transects were sampled monthly in the same manner as described for undisturbed transects. The effect that seasonality of denuding had on recolonization was determined in a second series of experiments that was established in September 1981, 30 months after the initial denuding, when all recolonization strips were reburned.

Exclusion cages

Four series of exclusion cage studies at each of the four recolonization stations were undertaken to determine the effects of grazing and predation on recolonization rates and patterns. The first series began in April 1979; nine areas were selected at each station, three areas in each of three tide zones (high, mid, and low tidal levels). In each area, two 20 x 20 cm patches were burned and cleared; one was covered with a stainless steel mesh cage (20 x 20 x 5 cm, 3 mm mesh), the second left as a control. Each month the percent cover of colonizing organisms was determined. The effect that season of denuding had on rates and patterns of recolonization was determined, as with the recolonization transect experiments. Subsequent series of exclusion cage experiments began in June 1980, September 1981, and December 1982; each area was reburned 15 months after the previous denuding.

The complete series of recolonization experiments (strip transects and exclusion cages) was completed under two-unit operating conditions by March 1984, but the recolonization strips were monitored bimonthly until March 1986 to assess long-term recovery. The observed degree of recolonization provides a base-line against which to compare the impact of Unit 3 on rates and patterns of community recovery. The entire series of experiments will be repeated under three-unit conditions.

Ascophyllum nodosum Studies

Growth and mortality of populations of the brown perennial alga, *Ascophyllum nodosum*, were studied at two control stations (GN, 5.5 km west of the discharge and WP, 1.5 km east of the discharge, Fig. 2) and an experimental station (FL, ca. 75 m east of the original Millstone quarry cut, Fig. 3) from 1979-1984. *Ascophyllum* was eliminated from FL in summer 1984, its loss attributed to elevated water temperatures resulting from the thermal plume of two operating units discharging through two quarry cuts (NUSCo 1985; also Appendix RS IV, Ecological Significance of Community Changes at Fox Island). In spring 1985 a second experimental *Ascophyllum* station (FN) was established between FE and FS (Fig. 3; ca. 250 m from the quarry discharges, northeast of the Fox Island-Exposed sampling site).

Ascophyllum plants were measured at monthly intervals from April, after the onset of new vesicle formation, until the following April. Fifty plants at each station were marked with a numbered plastic tag at the base of each plant, and five apices were marked on each plant with colored cable ties (prior to

1985, with colored electrical tape on uncolored cable ties). Linear growth was determined by measurements made from the top of the most recently formed vesicle to the apex of the developing axis, or apices if branching had occurred. Vesicles had not developed sufficiently to be tagged in April or May, so five tips were measured on each of 50 randomly chosen *Ascophyllum* plants. Monthly measurements of tagged plants began in June. Lost tags were not replaced, and the pattern of loss was used as a measure of both mortality and environmental stress. Loss of the entire plant was assumed when the base tag and tip tags were missing. Tip survival was determined in terms of remaining tip tags.

Temperature

Water temperatures were derived from the EDAN (Environmental Data Acquisition Network) system, which continuously records a variety of environmental parameters and reports at 15-minute intervals. Ambient water temperatures were recorded by sensors in Unit 1 and 2 intake bays, and effluent water temperatures by sensors in the quarry cuts (Fig. 3). Temperatures at FE and the experimental *Ascophyllum* stations (FL and FN, Fig. 3) were interpolated, based on measured ΔT, and verified over several tidal cycles with a portable thermistor and strip chart recorder.

Data Analysis

Relative abundance of intertidal organisms was estimated on the basis of percentage of substratum covered by each taxon. Unoccupied substrata were classed as free space. Similarity between communities was determined by a percent standardized form of the Bray-Curtis coefficient (Sanders 1960), calculated as:

$$S_{jk} = \sum_{i=1}^{n} \min(P_{ij}, P_{ik})$$

where P(ij) is the percent of species (i) at station (j), P(ik) is the percent for station (k), and (n) is the number of species in common. A flexible-sorting, clustering algorithm was applied to the resulting similarity matrix. The calculations were performed on untransformed percentages.

Ascophyllum growth data are reported as mean growth ± 2 standard errors; means are compared using 2-sample t-tests (PROC TTEST, SAS Institute Inc. 1982). A probability level of $\alpha = 0.05$ was used to

determine statistical significance. *Ascophyllum* mortality data are presented as the mean of years 1979-1986, with vertical bars representing the range of values. As a special case, mortality at Fox Island excludes 1984 data, because the elimination of *Ascophyllum* from FL in late summer of 1984 would bias comparisons between Fox Island and other stations.

RESULTS AND DISCUSSION

Temperature Data

Ambient water temperature in the Millstone Bight area follows a predictable annual cycle. Maximum daily average temperatures of 20-21 °C occur in August-September, with little variability among years. Winter temperatures are more variable, ranging from 0 to -1 °C in a cold year (e.g., 1977), to 3-4 °C in a warm year (e.g., 1983); yearly minima occur in January-February.

Condenser cooling water used by Millstone Units 1 and 2 is heated to 12-15 °C above ambient, depending on reactor power level. Prior to August 1983, the heated effluent was discharged through a single cut in the south end of the discharge quarry (Fig. 3); a second quarry cut was opened in anticipation of the added volume of cooling water needed by Unit 3.

With one cut open and either 1 or 2 units operating, the effluent plume was directed to Twotree Channel where it was subjected to tidal flushing. Water temperatures within 75 m of the cut (FL, *Ascophyllum* station; Fig. 3) were 2-3 °C above ambient, and temperatures at FE were within 1 °C of ambient regardless of tidal stage (Fig. 4, regime 1).

After the second discharge cut was opened the effluent plume lost half its momentum and mixed with nearshore water, producing nearly isothermal temperatures along the shore between the cuts and the southwest tip of Fox Island. During this time temperatures at Fox Island were 3-4 °C cooler than the undiluted effluent (Fig. 4, regime 2), regardless of tidal stage. This resulted in temperatures at FE 7-9 °C above ambient when one Unit was operating (e.g., autumn of 1983), and 12-13 °C above ambient when both Units were operating (e.g., most of 1984). Under these conditions, maximum water temperatures at FE could exceed 30 °C.

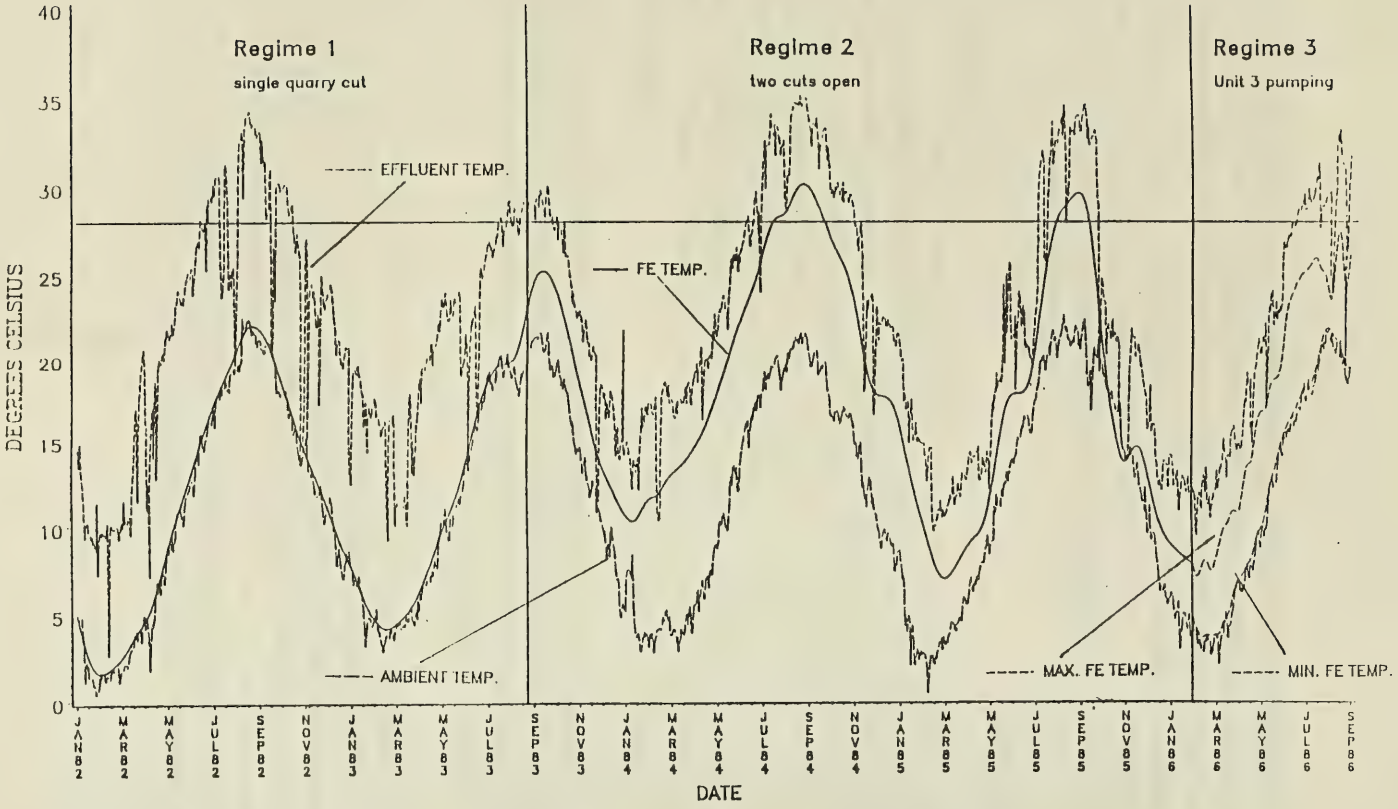

Figure 4. Water temperatures near MNPS, 1982-1986, showing results of second quarry cut opening: regime 1 = discharge through a single cut, regime 2 = discharge through two cuts, prior to start-up of Unit 3, and regime 3 = discharge through two cuts, after start-up of Unit 3.

With three units pumping, the effluent plume was again directed into Twotree Channel where it was subjected to tidal flushing. On the flood stage of the tide the effluent was directed westward, away from Fox Island. During high tide, ambient water temperatures were recorded at FE, and temperatures at FL were only 3-4 °C above ambient. On the ebbing tide, the plume was directed eastward across Fox Island. Maximum temperatures, up to 8-9 °C above ambient, occurred at FE and FL at low tide. Minimum temperatures were recorded 3 hours out of the 12 hour tidal cycle; elevated temperatures occurred during the remainder of the 12 hour cycle. These conditions are modelled in Figure 5 and produce regime 3 temperatures illustrated in Figure 4 (note two curves for temperature at FE in this period). Water temperatures are dependent on both the number of units operating and cooling water flow, and only when all units are pumping will there be a tidal component to the temperature. Therefore, conditions are likely to vary from year to year as each unit undergoes scheduled and unscheduled shut-downs. The effects of these changing conditions will be examined in subsequent reports. Most of the data presented in this report were collected under conditions modelled as regimes 1 and 2.

Qualitative Studies

Qualitative studies were designed to identify algal species present in intertidal and shallow subtidal areas in the vicinity of MNPS throughout the year, and to characterize their patterns of spatial and temporal distribution. Changes in these patterns indicate environmental changes, and suggest a close analysis of whether the changes were related to construction or operation of MNPS.

A rich and diverse flora occupies the rocky intertidal monitoring area, relative to other areas of Long Island Sound. Overall, 158 algal species have been identified since 1979. Not all species were found at any one station, nor were they all found in any one collection period, nor in any one year. Qualitative algal collections for the monitoring period 1979-1985 are presented as number of times each species was found in each month and at each station (Table 1). Complete collection records for each station are presented in Appendix RS Ia.

The benthic flora of the Greater Millstone Bight can be separated into five divisions (sensu Whittaker 1969): Rhodophyta (reds), Phaeophyta (browns), Chlorophyta (greens), Cyanophyta (blue-greens), and Chrysophyta (golden and yellow-green, including diatoms). In this study, blue-greens and diatoms are not identified to lower taxa. The benthic algal flora can also be classified on the basis of life history, e.g.,

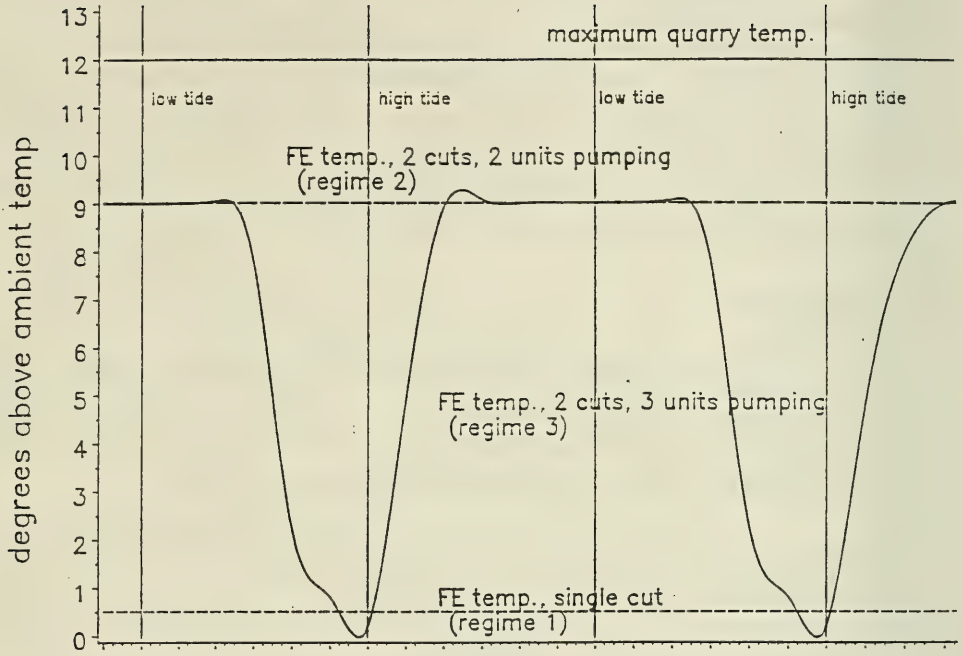

Figure 5. Idealized model of water temperatures at Fox Island-Exposed, related to number of quarry cuts and number of units operating. Regimes refer to Figure 4.

Table 1. Number of times each species was collected, by month and station, all years combined (1979-1985).

Rhodophyta	GN	BP	MP	TT	FE	FS	WP	SE	SS	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	TOT *
Goniotrichum alsidii	12	9	3	1	10	9	8	1	5	3	1	1	1	.	2	5	7	12	19	4	3	58
Erythrotrichia ciliaris	27	17	9	12	21	17	23	9	19	18	10	10	7	9	6	6	10	18	27	19	14	154
Erythrotrichia carnea	3	2	.	1	1	3	3	3	.	2	2	.	.	1	1	.	.	5	2	2	1	16
Erythrocladia subintegra	1	3	2	1	3	.	.	1	2	.	.	1	.	.	1	.	1	1	3	3	3	13
Erythropeltis discigera	3	5	3	.	4	2	1	2	4	4	2	1	.	2	.	1	1	4	3	3	3	24
Bangia atropurpurea	40	38	29	19	35	25	41	32	21	38	46	44	49	15	6	2	4	10	13	21	32	280
Porphyra leucosticta	35	23	30	23	27	19	31	26	20	27	41	35	37	25	15	7	5	5	13	10	14	234
Porphyra umbilicalis	54	35	25	29	45	35	44	54	28	27	30	44	44	53	41	33	23	14	8	15	17	349
Porphyra linearis	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	1	1
Porphyropsis coccinea	.	.	.	.	.	2	.	.	.	1	1	.	.	.	.	.	.	.	.	.	.	2
Audouinella purpurea	2	.	.	.	12	.	1	.	1	2	2	.	2	1	1	1	2	1	.	3	1	16
Audouinella secundata	33	41	22	13	20	20	25	23	21	20	31	21	20	20	23	14	13	12	22	11	11	218
Audouinella daviesii	4	3	1	3	2	2	3	3	3	5	.	1	4	2	2	3	1	1	1	4	.	24
Audouinella saviana	14	10	9	7	10	11	7	2	5	9	3	8	7	5	6	3	3	11	10	4	6	75
Audouinella sp.	1	1	.	.	.	.	.	.	.	.	.	.	.	.	.	2	.	.	.	.	.	2
Audouinella dasyae	.	2	.	.	.	.	.	.	.	.	.	.	.	.	1	.	1	.	.	.	.	2
Gelidium crinale	6	.	.	2	1	53	1	.	.	4	5	2	5	4	5	6	5	5	8	7	7	63
Bonnemaisonia hamifera	1	17	1	17	.	1	17	24	28	5	6	10	10	15	30	19	5	.	.	3	3	106
'Trailliella intricata'	.	.	.	.	.	.	.	.	1	.	.	1	.	.	.	.	.	.	.	.	.	1
Agardhiella subulata	4	6	4	1	20	7	14	3	10	3	3	3	2	6	8	7	7	10	6	7	7	69
Polyides rotundus	3	7	4	4	4	7	15	2	22	3	6	.	4	6	2	6	7	8	8	11	7	68
Cystoclonium purpureum	49	42	36	35	49	49	60	41	52	48	46	39	42	45	45	26	13	7	27	34	41	413
Gracilaria tikvahiae	.	1	.	.	1	.	.	.	.	.	.	.	.	.	.	.	1	1	.	.	.	2
Ahnfeltia plicata	17	33	31	49	61	16	44	20	46	29	30	28	28	24	25	21	21	21	27	29	34	317
Phyllophora pseudoceranoides	7	6	8	17	8	7	34	6	9	15	9	9	9	6	6	7	7	2	11	11	10	102
Phyllophora truncata	9	9	7	6	9	9	18	4	11	7	12	4	3	5	7	7	5	6	6	9	11	82
Chondrus crispus	84	84	53	53	66	84	84	84	84	57	57	56	56	56	56	56	56	55	57	57	57	676
Gigartina stellata	21	51	47	53	14	18	55	81	82	44	34	30	27	32	33	34	32	33	38	46	39	422
Petrocelis middendorfii	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	1	1
Rhodophysema georgii	.	1	.	.	.	2	3	1	10	.	1	4	1	5	1	1	2	2	.	.	.	17
Corallina officinalis	2	84	53	19	80	59	69	27	23	36	36	33	32	29	29	35	36	34	41	35	40	416
Dumontia contorta	33	16	20	26	22	38	28	25	33	28	38	45	47	45	27	3	1	1	.	1	5	241
Gloisiphonia capillaris	.	.	1	.	1	.	.	.	.	1	.	1	.	.	.	.	.	.	.	.	.	2
Choreocolax polysiphoniae	12	18	2	1	2	.	4	.	5	5	7	7	5	5	1	2	2	1	3	1	5	44
Hildenbrandia rubra	1	.	.	8	.	1	1	.	1	2	1	1	1	.	.	1	.	1	3	1	1	12
Palmaria palmata	30	33	9	37	7	7	37	21	49	19	20	28	25	22	26	19	16	12	7	18	18	230
Champia parvula	31	29	9	19	26	32	51	22	39	20	14	6	4	2	2	20	37	42	46	38	27	258
Lomentaria baileyana	18	6	1	4	14	11	14	2	2	2	.	.	.	.	.	3	17	28	16	5	1	72
Lomentaria clavellosa	4	4	1	7	2	4	8	6	10	6	5	5	9	4	1	1	2	4	2	4	3	46
Lomentaria orcadensis	1	.	.	4	.	.	2	.	1	1	1	2	.	.	.	.	.	1	3	.	.	8
Antithamnion americanum	1	1	2	1	.	.	4	.	1	1	1	1	1	2	.	.	.	1	2	1	.	10
Antithamnion cruciatum	37	44	18	21	28	34	52	25	36	27	14	3	10	4	9	26	36	40	43	43	40	295
Antithamnion pylaisaei	.	2	.	.	.	1	1	1	1	2	.	.	.	1	.	.	1	.	.	1	1	6
Antithamnion sp.	.	.	1	.	.	.	.	.	1	.	.	.	1	.	.	.	.	.	.	.	1	2
Callithamnion corymbosum	.	6	1	1	2	1	4	1	1	.	.	.	.	.	1	.	4	5	4	3	.	17
Callithamnion roseum	8	5	5	3	15	9	8	3	7	4	1	1	.	.	.	5	10	20	10	8	4	63
Callithamnion tetragonum	32	31	30	35	42	21	38	19	26	39	28	13	19	12	6	14	15	25	28	43	32	274

Table 1. (cont).

Rhodophyta	GN	BP	MP	TT	FE	FS	WP	SE	SS	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	TOT
Callithamnion byssoides	1	4	.	.	.	.	.	.	.	.	.	.	.	.	.	2	2	1	.	.	.	5
Callithamnion baileyi	2	2	1	1	.	.	1	.	1	.	.	.	.	.	.	.	.	.	3	4	1	8
Ceramium deslongchampii v hooperi	11	1	.	2	.	3	1	1	.	2	2	.	1	.	1	.	2	2	2	2	5	19
Ceramium diaphanum	15	24	1	16	7	11	22	16	16	4	.	.	1	.	.	14	39	28	29	6	7	128
Ceramium rubrum	74	80	47	49	73	65	79	73	72	52	52	42	46	51	52	54	48	54	55	54	52	612
Ceramium fastigiatum	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	.	1	.	.	.	1
Spermothamnion repens	39	49	23	25	18	23	57	22	40	32	20	16	15	10	16	16	20	23	42	44	42	296
Spyridia filamentosa	10	1	.	.	.	3	1	.	.	.	.	.	.	.	1	1	2	7	2	1	1	15
Grinnellia americanum	1	1	1	2	1	1	11	2	10	2	.	.	1	1	.	2	5	1	6	7	5	30
Rhycodrys rubens	.	.	1	5	1	2	15	5	9	1	3	3	7	4	4	2	3	3	4	4	.	38
Dasya baillouviana	7	12	2	6	9	18	15	3	11	4	1	.	.	.	.	4	22	17	15	14	6	83
Chondria sedifolia	4	1	.	1	.	1	.	.	1	.	.	.	.	.	.	.	2	3	3	.	.	8
Chondria baileyana	3	4	6	2	.	3	3	.	1	1	.	1	.	1	.	.	2	9	5	.	3	22
Chondria tenuissima	4	.	.	.	1	1	.	.	.	.	.	.	1	.	.	2	1	2	.	.	.	6
Chondria dasyphylla	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	1	.	.	.	1
Polysiphonia denudata	.	3	.	2	4	2	5	1	1	.	.	.	.	.	.	3	.	2	5	4	4	18
Polysiphonia harveyi	49	53	29	29	59	45	50	43	48	42	24	14	13	11	30	47	54	52	39	41	38	405
Polysiphonia lanosa	72	72	53	24	39	23	68	42	79	49	43	39	34	40	36	35	34	37	42	39	44	472
Polysiphonia nigra	1	14	1	1	3	6	10	5	9	3	5	4	6	10	9	1	2	1	3	2	4	50
Polysiphonia nigrescens	16	19	3	8	13	17	46	8	17	11	8	10	18	11	16	13	9	10	16	13	12	147
Polysiphonia urceolata	35	17	11	13	20	4	24	3	11	11	9	17	31	34	18	10	2	2	1	1	2	138
Polysiphonia elongata	.	.	.	1	.	.	4	.	.	.	.	.	.	.	1	.	1	.	.	1	2	5
Polysiphonia fibrillosa	4	1	1	4	3	.	2	1	1	4	1	.	.	.	.	1	.	.	2	.	9	17
Polysiphonia flexicaulis	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	.	1	1
Polysiphonia novae-angliae	24	30	31	26	31	25	32	21	27	29	22	11	10	12	11	18	19	20	32	30	33	247
Rhodomela confervoides	12	7	1	6	4	2	5	4	8	4	5	17	11	5	2	.	1	1	1	1	1	49

Phaeophyta	GN	BP	MP	TT	FE	FS	WP	SE	SS	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	TOT
Ectocarpus fasciculatus	14	17	25	18	23	8	17	17	13	4	10	7	15	19	20	14	9	16	20	12	6	152
Ectocarpus siliculosis	41	32	13	27	37	31	38	29	24	11	19	27	30	34	40	34	22	16	16	14	9	272
Ectocarpus sp.	2	5	8	6	2	3	2	3	7	3	7	5	4	.	3	4	2	4	1	3	2	38
Giffordia granulosa	2	3	4	3	4	2	2	1	3	3	1	2	2	2	4	1	.	3	1	3	2	24
Giffordia mitchelliae	15	11	3	4	27	13	21	5	5	2	4	1	3	8	11	11	18	18	16	8	4	104
Pilayella littoralis	55	6	3	10	5	49	17	6	5	12	10	14	20	29	18	7	9	8	12	7	10	156
Spongonema tomentosum	14	17	6	8	10	4	9	11	9	10	17	24	17	11	.	1	2	.	4	1	1	88
Entonema aecidioides	.	2	.	.	.	.	.	1	.	.	.	.	2	1	.	.	.	.	.	.	.	3
Acinetospora sp.	.	1	.	.	.	2	.	1	.	.	.	.	2	1	.	.	1	.	.	.	.	4
Feldmannia sp.	.	1	.	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	1	1	.	2
Ralfsia verrucosa	64	49	22	9	51	50	57	23	35	26	33	21	22	26	27	33	37	39	38	31	27	360
Elachista fucicola	59	52	35	30	51	38	51	61	45	24	31	35	41	49	47	40	46	44	28	21	16	422
Halothrix lumbricalis	1	3	.	.	.	.	3	.	2	.	1	.	.	3	2	1	.	1	1	.	.	9
Leathesia difformis	10	1	13	2	12	1	10	1	3	.	1	.	5	11	15	15	6	.	.	.	.	53
Chordaria flagelliformis	13	19	13	7	4	2	23	10	6	.	.	.	3	11	28	21	17	11	3	1	2	97
Sphaerotrichia divaricata	.	1	1	2	.	.	3	3	1	.	.	.	1	3	6	1	.	.	.	.	.	11
Eudesme zosterae	.	1	.	.	.	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	1
Asperococcus fistulosus	3	1	2	2	.	2	4	1	1	.	8	3	1	3	.	.	.	1	.	.	.	16

Table 1. (cont).

Phaeophyta	GN	BP	NP	TT	FE	FS	WP	SE	SS	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	TOT
Desmotrichum undulatum	6	5	.	.	.	2	4	.	3	.	1	8	4	2	1	.	.	.	2	.	2	20
Phaeosaccion collinsii	1	.	1	.	.	1	1	.	.	.	2	1	.	.	.	.	.	.	1	.	.	4
Punctaria latifolia	4	4	1	5	2	9	4	4	3	1	5	9	7	2	5	.	.	.	3	2	2	36
Punctaria plantaginea	4	3	2	.	.	6	6	.	1	1	1	2	1	3	4	4	3	2	.	1	.	22
Petalonia fascia	45	62	34	25	45	47	59	46	32	41	50	39	49	48	48	41	5	3	7	26	38	395
Scytosiphon lomentaria	54	62	26	26	41	54	50	41	34	28	47	53	54	53	54	49	10	3	5	12	20	388
Delamarea attenuata	.	.	1	.	.	1	.	.	.	.	1	1	.	.	.	.	.	.	.	.	.	2
Desmarestia aculeata	2	1	2	10	4	8	14	4	9	5	2	5	9	5	7	1	4	2	5	7	2	54
Desmarestia viridis	10	14	8	13	7	9	14	7	16	1	3	17	25	28	22	1	.	.	.	1	.	98
Chorda filum	2	1	.	6	1	1	9	3	3	.	.	.	1	5	11	9	.	.	.	.	.	26
Chorda tomentosa	.	3	1	11	2	3	5	3	7	.	.	3	10	16	1	5	.	.	.	.	.	35
Laminaria digitata	.	.	.	7	.	.	.	.	.	1	1	.	.	2	.	.	1	.	.	.	2	7
Laminaria longicruris	9	3	6	22	1	1	16	12	12	5	7	5	8	8	6	7	6	7	10	8	5	82
Laminaria saccharina	52	58	33	51	50	42	51	52	45	32	22	30	36	47	44	47	43	34	35	30	34	434
Sphacelaria cirrosa	41	21	6	3	41	23	15	.	3	18	9	5	7	9	9	11	12	12	18	21	22	153
Sphacelaria furcigera	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	1	.	.	.	1
Ascophyllum nodosum	84	84	53	53	66	84	84	84	84	57	57	56	56	56	56	56	56	55	57	57	57	676
Fucus distichus s edentatus	5	6	8	11	9	5	2	6	1	5	7	10	12	9	2	1	1	.	1	3	2	53
Fucus distichus s evanescens	9	10	1	13	5	5	7	8	10	9	7	12	11	13	4	1	3	.	2	6	.	68
Fucus spiralis	4	28	5	.	.	1	3	2	1	1	1	1	5	3	6	4	4	8	4	5	2	44
Fucus vesiculosus	84	84	53	53	69	84	84	84	84	57	57	56	56	56	57	57	57	55	57	57	57	679
Sargassum filipendula	.	1	.	.	.	.	.	.	.	.	.	.	.	.	.	.	1	.	.	.	.	1

Chlorophyta	GN	BP	NP	TT	FE	FS	WP	SE	SS	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	TOT
Ulothrix flacca	31	31	15	9	27	24	38	26	20	31	36	42	40	27	4	3	.	1	5	14	18	221
Urospora penicilliformis	32	35	22	17	26	18	36	26	20	36	42	42	40	17	4	1	2	1	4	17	26	232
Urospora wormskjoldii	4	7	3	.	6	3	6	2	.	5	4	5	9	1	4	2	.	.	.	.	1	31
'Urospora collabens'	5	5	3	2	3	1	2	2	5	4	11	5	1	3	1	1	.	.	.	.	2	28
Entocladia viridis	.	2	3	.	.	1	.	.	.	1	1	.	1	.	.	.	.	.	1	1	1	6
Monostroma grevillei	22	19	9	10	19	27	19	13	21	14	34	31	34	29	7	1	1	1	.	4	3	159
Monostroma pulchrum	27	28	13	15	20	19	28	28	26	11	27	50	52	49	9	2	.	.	2	1	1	204
Spongomorpha arcta	24	23	15	12	15	3	15	8	6	4	10	22	29	31	18	3	.	.	.	2	2	121
Spongomorpha aeruginosa	8	4	7	.	5	5	7	4	1	2	1	4	8	10	9	3	2	2	.	.	.	41
'Codiolum gregarium'	1	.	.	.	.	.	.	.	.	.	.	.	.	.	.	1	.	.	.	.	.	1
Capsosiphon fulvescens	1	1	.	.	.	2	5	.	2	.	1	.	3	2	2	1	.	2	.	.	.	11
Blidingia minima	51	46	35	36	59	7	54	36	52	33	25	25	30	40	37	27	38	33	29	28	31	376
Blidingia marginata	1	2	.	1	3	1	3	1	2	5	2	.	.	1	.	2	.	.	2	1	1	14
Enteromorpha clathrata	23	3	5	2	21	28	28	1	12	2	3	1	7	10	11	21	26	21	16	3	2	123
Enteromorpha flexuosa	32	34	18	15	53	31	45	10	28	19	19	14	16	21	23	23	18	26	35	32	20	266
Enteromorpha groenlandica	5	17	4	2	2	2	4	2	4	1	8	7	8	6	4	1	.	.	.	1	6	42
Enteromorpha intestinalis	44	33	19	9	27	23	46	13	25	14	15	20	25	28	27	26	29	20	14	12	9	239
Enteromorpha linza	42	62	40	21	60	29	56	40	35	30	18	18	32	38	36	37	34	34	42	37	29	385
Enteromorpha prolifera	35	31	21	19	29	29	45	13	34	25	22	19	16	20	21	14	19	18	28	26	28	256
Enteromorpha torta	4	1	1	.	1	5	5	.	2	1	.	.	.	2	4	3	4	1	3	1	.	19
Enteromorpha ralfsii	3	1	.	.	3	7	3	.	.	.	.	.	.	.	6	4	2	3	1	1	.	17
Percursaria percursa	3	1	.	.	1	7	3	.	1	1	.	.	3	2	5	2	1	1	1	.	.	16

Table 1. (cont).

Chlorophyta	GN	BP	MP	TT	FE	FS	WP	SE	SS	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	TOT
Ulva lactuca	81	76	50	50	81	75	82	72	72	57	52	44	48	51	56	53	52	55	58	57	56	639
Ulvaria oxysperma	.	1	.	.	.	.	.	.	2	1	.	.	1	1	.	.	.	.	.	.	.	3
Prasiola stipitata	43	1	2	48	1	1	.	63	7	12	13	13	13	13	13	16	14	14	15	12	18	166
Chaetomorpha linum	47	68	46	43	58	57	63	66	71	47	35	23	21	32	51	52	54	55	55	50	44	519
Chaetomorpha melagonium	1	.	.	.	.	.	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	1
Chaetomorpha aerea	19	22	14	2	62	34	38	2	8	17	14	15	12	11	15	21	16	24	18	21	17	201
Cladophora albida	4	3	2	.	5	7	12	2	5	.	.	1	1	6	5	9	6	7	2	2	1	40
Cladophora flexuosa	7	22	15	8	11	9	16	14	10	8	1	2	4	8	14	21	15	10	16	7	6	112
Cladophora glaucescens	.	.	1	.	.	.	3	.	.	.	.	.	.	1	1	1	1	.	.	.	.	4
Cladophora laetevirens	.	.	1	.	.	1	2	1	4	.	.	.	1	1	2	2	.	2	1	.	.	9
Cladophora refracta	7	31	13	4	15	10	20	14	10	5	4	1	1	3	18	19	15	16	22	10	10	124
Cladophora sericea	20	17	6	1	34	27	34	9	17	7	3	3	14	30	24	20	21	13	12	10	8	165
'Cladophora crystallina'	.	.	.	.	.	.	.	2	.	.	.	.	.	.	1	1	.	.	.	.	.	2
Cladophora hutchinsiae	4	1	5	2	11	6	10	2	9	1	2	2	1	5	5	4	7	7	9	6	1	50
Cladophora rupestris	2	2	1	2	2	3	3	.	2	.	1	.	1	3	1	5	4	.	1	1	.	17
Cladophora ruchingeri	.	.	.	.	1	.	.	.	.	.	.	.	.	.	.	.	1	.	.	.	.	1
Rhizoclonium riparium	24	16	4	3	16	41	24	8	8	5	17	14	9	10	10	17	17	16	13	7	9	144
'Rhizoclonium kerneri'	2	1	.	.	.	1	.	1	1	1	.	2	.	.	2	.	.	.	.	.	1	6
Rhizoclonium tortuosum	1	.	.	.	2	.	1	.	.	.	.	.	.	.	3	.	.	.	.	1	.	4
Bryopsis plumosa	6	4	.	4	16	3	4	5	3	4	.	.	.	2	1	7	9	6	7	5	4	45
Bryopsis hypnoides	2	3	2	.	3	5	1	1	3	.	.	.	.	1	3	5	1	3	4	1	2	20
Derbesia marina	5	2	.	3	21	6	.	.	2	4	4	2	5	.	.	1	2	2	5	7	7	39
Codium fragile	68	71	47	53	84	73	68	54	58	53	47	39	43	43	47	46	48	50	56	53	51	576

* out of a possible 684 collections

perennials, pseudoperennials, seasonal annuals, and aseasonal annuals. Perennials (species present as a whole thallus throughout the year and persisting for more than one year) include *Chondrus crispus*, *Ascophyllum nodosum*, and *Fucus vesiculosus*. Pseudoperennials (individual plants present throughout the year but passing through adverse conditions in a reduced form) include *Codium fragile*. Seasonal annuals (plants only found during part of the year) include *Bangia atropurpurea*, *Desmarestia viridis*, *Cladophora flexuosa*, and *Sphacelaria cirrosa*. Aseasonal annuals (population as a whole present throughout the year and capable of continual reproduction) include *Ulva lactuca* and *Enteromorpha flexuosa*.

Of the 158 species of benthic algae found in our area, several were site-specific or characteristic of only one station (Table 1), such as *Laminaria digitata* at TT, *Ceramium deslongchampii* at GN, *Fucus spiralis* at BP, *Bryopsis plumosa* at FE, *Polysiphonia nigrescens* at WP, and *Gelidium crinale* at FS. In addition, some species are abundant at only 2 or 3 stations, e.g., *Pilayella littoralis* at GN and FS, and *Prasiola stipitata* at GN, TT, and SE.

Temporal differences also occur, and seasonal components in the local flora have been identified. Some examples include *Bangia atropurpurea*, *Dumontia contorta*, and *Monostroma grevillei* as most common in winter-spring, *Desmarestia viridis*, *Leathesia difformis*, and *Polysiphonia urceolata* in spring-summer, *Champia parvula*, *Dasya baillouviana*, and *Cladophora flexuosa* in summer-autumn, and *Callithamnion tetragonum*, *Spermothamnion repens*, and *Sphacelaria cirrosa* in autumn-winter (Table 1).

The persistence of these spatial and temporal patterns, and their consistency from year to year is an indication of the stability of the local flora. Another measure of stability is number of species, i.e., species richness.

The greatest number of species recorded from a single collection usually occurred in spring-early summer for all stations. The most species collected in any month (1979-1985) was 117 in July (Table 2), and the fewest was 101 in March. The most species collected at any station since 1979 was 131 at BP (Table 3), and the least was 109 at both SE and TT. In each year, the greatest species number generally occurred at WP (Table 4).

When division proportions are analyzed by month and station, proportions are similar and independent of species number (Tables 2 and 3), providing another measure of floral stability. Annual percentage of

Table 2. Number of species collected in each division, by month (1979-1985 combined), and their percentage (in parentheses).

Division	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Rhodophyta	53(48)	47(44)	45(45)	45(41)	43(38)	45(40)	50(43)	56(51)	59(54)	53(49)	54(49)	56(51)
Phaeophyta	24(22)	30(28)	28(28)	32(29)	34(30)	29(26)	28(24)	25(23)	22(20)	26(24)	25(23)	23(21)
Chlorophyta	33(30)	31(29)	28(28)	33(30)	36(32)	38(34)	39(33)	29(26)	29(26)	30(28)	32(29)	31(28)
total	110	108	101	110	113	112	117	110	110	109	111	110

Table 3. Number of species collected in each division, by station (1979-1985 combined), and their percentage (in parentheses).

Division	GN	BP	MP	TT	FE	FS	WP	SE	SS	total
Rhodophyta	56(45)	57(44)	51(45)	54(50)	50(45)	54(44)	58(46)	48(44)	57(46)	73(46)
Phaeophyta	30(24)	36(27)	30(27)	28(26)	26(23)	32(26)	33(26)	29(27)	31(25)	40(25)
Chlorophyta	39(31)	38(29)	32(28)	27(25)	36(32)	38(31)	36(28)	32(29)	36(29)	45(28)
total	125	131	113	109	112	124	127	109	124	158

Table 4. Number of species at each station in each year (1979-1985), yearly totals, and yearly division counts and percentages (in parentheses). Each 'year' represented by collections from March to following February.

Station	1979	1980	1981	1982	1983	1984	1985
BP	72	70	94	86	80	85	73
FE	76	72	77	80	70	65	50
FS	71	69	72	80	79	82	65
GN	67	74	85	85	82	84	77
MP*	-	-	-	75	77	77	74
SE	58	55	71	73	69	70	63
SS	67	67	84	88	85	85	68
TT*	-	-	-	87	80	77	78
WP	72	82	96	95	90	90	87
total	100	103	129	131	126	133	117
Division							
Rhodophyta	44(44)	47(46)	60(47)	58(44)	57(45)	61(46)	54(46)
Phaeophyta	26(26)	26(25)	35(27)	35(27)	32(26)	34(25)	29(25)
Chlorophyta	30(30)	30(29)	34(26)	38(29)	37(29)	38(29)	34(29)

species in each division (1979-1985) ranged from 44-47% for reds, browns 25-27%, and greens 26-30% (Table 4), but the proportions of reds, browns, and greens vary seasonally (Table 2). Reds are more prevalent, but especially so in August-December; browns are less prevalent than reds or greens, but have their highest percentages in February-May; greens are proportionally most common in May-July. The overall number of species in each phylum is 73 reds, 40 browns, and 45 greens (all stations, all years); their proportion is 46:25:29 (Table 4).

The local flora, when represented as percent occurrence of reds, browns, and greens, is consistent with those reported by other researchers in the northwest Atlantic (Vadas 1972; Schneider et al. 1979; Mathieson et al. 1981; Mathieson and Hehre 1986). Vadas (1972) reported the proportion 45:32:23 on an open coast in Maine. Mathieson et al. (1981), working in the Great Bay estuary system and adjacent open coast of New Hampshire-Maine, reported the proportion 47:28:25, and Mathieson and Hehre (1986) updated this study reporting a New Hampshire open coast proportion of 43:31:26. In a checklist of Connecticut algae (including intertidal and subtidal species), Schneider et al. (1979) reported a proportion of 45:26:29, which included 188 species, not including varieties and forms. Proportionally, the Connecticut marine flora as determined by Schneider, is virtually identical to our own.

Exceptions to spatial and temporal trends are evident in the Fox Island-Exposed qualitative collections after the opening of the second quarry cut in August 1983. The average number of species per monthly collection was 31 from March 1979 to August 1983, and dropped to 28 in the year following the opening of the second cut. A sharper drop in species richness occurred after August-September 1984. An average of 18 species per month was collected from September 1984 to February 1986, and in October 1984, only 10 species of algae were collected from FE (the collection with lowest number of species from any station in any month since the beginning of the study). Species richness decreased because physiological limits of many species were exceeded; water temperatures at Fox Island exceeded 28 °C in August-September 1984 (NUSCo 1985; also Appendix RS IV, Ecological Significance of Community Changes at Fox Island). Community changes resulting from elevated water temperatures included the loss of established populations of perennial macroalgae (*Chondrus crispus*, *Fucus vesiculosus*, *Ascophyllum nodosum*) and associated epiphytes, and increased abundance and persistence of opportunistic species (*Codium fragile*, *Enteromorpha* spp., *Polysiphonia* spp.). The changes were evidenced both as a loss of species, and a shift in relative divisional proportions. From March 1979-August 1983 (prior to the second cut opening), 105 species were reported at FE; 70 species were reported from September 1983-August 1984, the first year after the second

cut opening, and 67 species were reported from September 1984-September 1986, beginning one year after the second cut opening. Proportions for the three periods were 43:25:32, 41:26:33, and 40:22:37, respectively; noticeable is the decrease in browns and reds with a concomitant increase in the proportion of greens. This shift in relative proportions in response to elevated water temperature was also seen in the quarry collections, and will be discussed in the next section.

The floristic changes noted at FE were localized, and not indicative of more widespread effects. Overall, the flora of Greater Millstone Bight has remained stable over time, and is similar to those studied elsewhere in New England by other researchers (Vadas 1972; Wilce et al. 1978; Schneider et al. 1979; Mathieson et al. 1981). The NUEL qualitative collections are important for predicting and assessing the impact of 3-unit operation. These studies permit the determination of the degree of variability in seasonal and yearly species occurrence. Qualitative differences in species composition among stations or years (as noted at FE after 1983), as compared to species composition at the reference sites, can signal power plant influence. Analyses of the flora in the years after Unit 3 becomes operational should indicate whether possible thermal effects will be within present bounds or will spread to other reference sites.

Quarry Study

Qualitative algal collections from the Millstone quarry permit characterization of sites exposed to a wide range of water temperatures, from ambient temperature when all reactors are shut down, to 21 °C above ambient, when all reactors are at full power. Water temperatures change in response to varying reactor power levels.

The overall quarry flora, composed of all species collected in the quarry or quarry cut, is similar but not identical, to that of the NUEL rocky shore sampling stations reported in the previous section. Of the 118 species collected from the quarry or quarry cut, only 3 have not been collected at other NUEL monitoring sites: *Audouinella flexuosa*, *A. sagraenum*, and *Sorocarpus micromorus*. Those species found at other sites but never in the quarry or the quarry cut are mostly cold-water reds, browns, and their epiphytes. The relative proportion of reds, browns, and greens at the MNPS quarry (1979-1986) was 46:21:33.

Some of the similarity between the quarry flora and that of the reference stations is due to periods of ambient water temperatures in the quarry as a result of periodic reactor shutdown. Schneider (1981)

sampled algae from the Millstone effluent quarry over an 18 month period (including several shutdowns) and found 42 species in the 3 major algal divisions, with proportions similar to those found in NUEL quarry studies (Table 5). All species noted by Schneider except *Spongomorpha arcta* were present in our quarry collections (this species was found in the quarry cut). Schneider also reported the range of temperatures over which each species occurred in his study area; relative algal proportions change with elevations in water temperature, a fact also evident in NUEL quarry data. In Schneider's study and our own, when temperatures exceeded 25 °C and 30 °C, a difference in species proportions became evident. With elevated temperatures, the number of brown algal species decreased rapidly in the quarry, thereby decreasing the proportion of browns relative to other groups. The relative proportions of species in each algal division are similar in both Schneider's study and the NUEL quarry study (Table 5). Divisional proportions are independent of species number and indicate phytogeographic affinity (Druehl 1981).

Table 5. Relative proportion of each major algal division (reds, browns, greens) from New England collections and MNPS quarry collections.

45:32:23	Maine open coast	Vadas (1972)
43:31:26	New Hampshire open coast	Mathieson and Hehre (1986)
47:28:25	Great Bay estuary and NH-Maine open coast	Mathieson *et al.* (1981)
45:26:29	CT intertidal and subtidal	Schneider *et al.* (1979)
46:25:29	reference sites	NUEL (1979-1986)
46:21:33	MNPS quarry	NUEL (1979-1986)
58:10:32	> 25 °C	
57: 3:40	> 30 °C	
45:24:31	MNPS quarry	Schneider (1981)
50:12:38	> 25 °C	
57: 5:38	> 30 °C	

These phytogeographic affinities are also evidenced in the temporal distribution of components of the quarry flora, relative to that of the reference sites. Species with southern centers of distribution (i.e., sub-tropical affinities), found mostly in summer and autumn in local rocky shore collections, had an extended growing season in the quarry. For example, *Agardhiella subulata* ocurred mostly in summer (May, June, July) at the rocky intertidal collection sites, but occurred in every month in the quarry effluent. Another red alga, *Dasya baillouviana*, was seasonal at reference stations (August-November) and found only 83 times since 1979. Over the course of the quarry study, *Dasya baillouviana* could be found in any month in the quarry. Similarly, *Enteromorpha clathrata* and *Cladophora sericea*, May-October greens at most rocky intertidal sites, were much more common in the quarry and over a longer period. After September 1984 at FE, *Agardhiella subulata*, *Enteromorpha clathrata*, and *Cladophora sericea*, among others, were nearly as common at FE as at the quarry site.

In contrast, algae with more northerly distributions (boreal affinities, especially browns) were less common in the quarry. *Laminaria saccharina* occurred each month at qualitative collection sites, but only in March and April at the quarry. Similarly, *Petalonia fascia*, a November-July brown at nearby coastal stations, occurred only in January and February at the quarry.

Qualitative algal collections from the MNPS effluent quarry are important because they characterize a flora exposed to water temperature higher than those found at any nearby station. This flora therefore reflects environmental conditions at one end of a thermal gradient; the floral characteristics of the reference stations reflect ambient coastal thermal regimes. Differences in the quarry flora, i.e., reduced number of species, especially browns, with increasing temperature, the resultant shift in divisional proportions, and the temporal displacement (extension or reduction of an alga's growing season) are responses to this thermal gradient. If the flora at a station farther from the discharge evidences similar floristic or vegetative change, particularly after Unit 3 begins operation, a critical standard has been developed against which thermal impact may be assessed.

Undisturbed Transects

One of the most noticeable biological features of local rocky shore communities is zonation; i.e., the segregation of intertidal organisms into horizontal bands, each characterized by a particular complex of

plants and animals. This phenomenon is considered to be a universal feature of rocky shores (Stephenson and Stephenson 1949, 1972; Lewis 1964).

Locally, we recognize three intertidal zones; identified as the high, mid, and low intertidal. The high intertidal (Zone 1), primarily bare rock, is seasonally occupied by barnacles (*Balanus balanoides*) and ephemeral algae (mostly *Ulothrix flacca*, *Bangia atropurpurea*, *Blidingia minima*, or blue-greens). *Fucus* (mostly *F. vesiculosus*, occasionally *F. spiralis*) may occur in small amounts in the high intertidal.

Barnacles and the fucoids *Fucus vesiculosus* and *Ascophyllum nodosum* dominate the mid intertidal (Zone 2). Other algae in Zone 2 grow directly on rock (e.g., *Ralfsia verrucosa*, *Enteromorpha* spp.) or as epiphytes of fucoids (e.g., *Elachista fucicola*, *Polysiphonia* spp.).

The low intertidal (Zone 3) is typically dominated by *Chondrus crispus*, though fucoids may also be common. Barnacles are seasonally abundant, but usually obscured by an algal canopy. Other algae may be attached to rock (e.g., *Ralfsia*, *Corallina officinalis*, *Dumontia contorta*) or to larger algae. *Monostroma pulchrum* and *Polysiphonia* spp. are common ephemeral epiphytes.

The preceding description is not meant to imply that local intertidal communities are static or homogenous; that is far from the case. Rather, the rocky shore is patchy, a mosaic of plants and animals in dynamic equilibrium. Organisms compete for space, light, and nutrients. Processes of recruitment, colonization, and growth are balanced against predation, senescence, and death. These processes vary over space and time, on varying scales.

General patterns of spatial and temporal distribution may be illustrated by plotting abundance (measured as percent substratum coverage) over time for the major components of the local intertidal communities (Figs. 6-10); data for all taxa are included in Appendix RS Ib. In each case, data are presented as: a) a time-series beginning in March 1979, to show long-term trends and year-to-year variability, and b) all years combined, calculating monthly means ± 2 SE, to show seasonal trends. For example, Figure 6 presents barnacle coverage in each zone, and abundance of predatory snails (mostly *Urosalpinx cinerea* and *Thais lapillus*) in Zone 3. Generally, barnacle coverage is highest in the mid intertidal (Zone 2) and higher at exposed stations (e.g., BP, FE) than at sheltered stations (e.g., SS, FS). There are differences between years, but most variability occurs annually, i.e., a seasonal cycle. Barnacles settle in early spring,

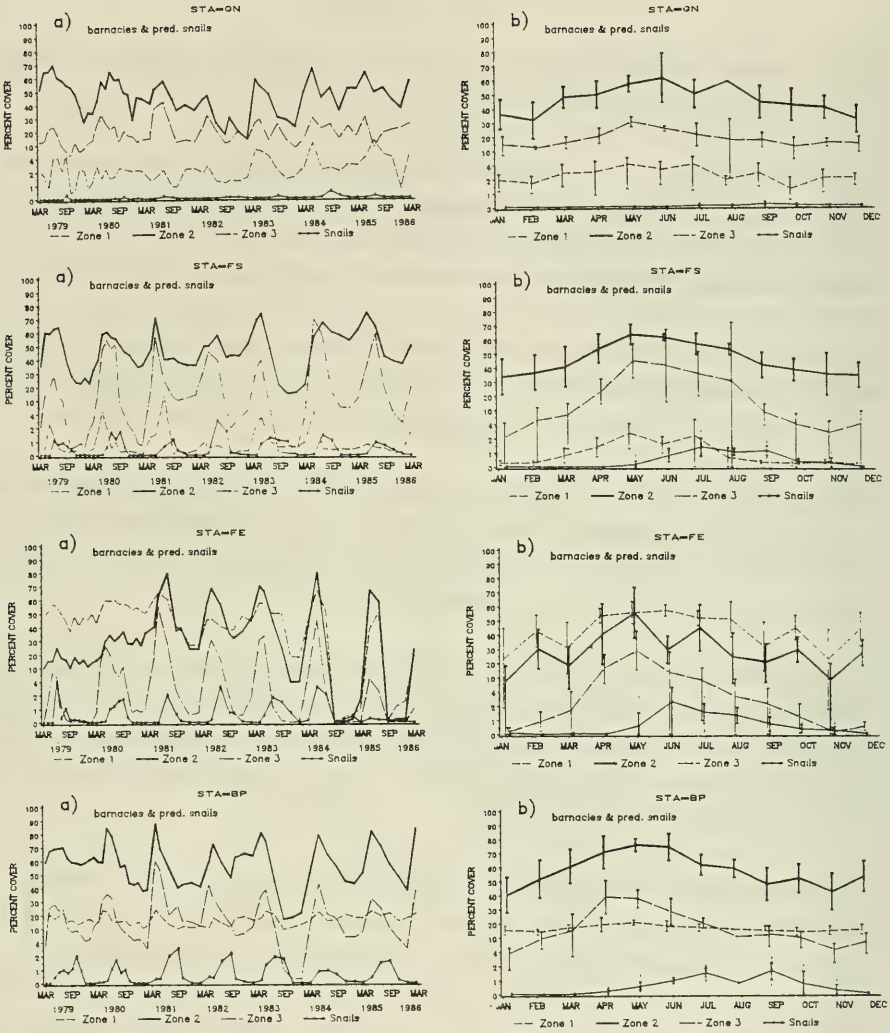

Figure 6. Abundance of barnacles and predatory snails as percent cover in each zone: (a) from March 1979-March 1986, and (b) monthly from 1979-1986, for GN, FS, FE, and BP.

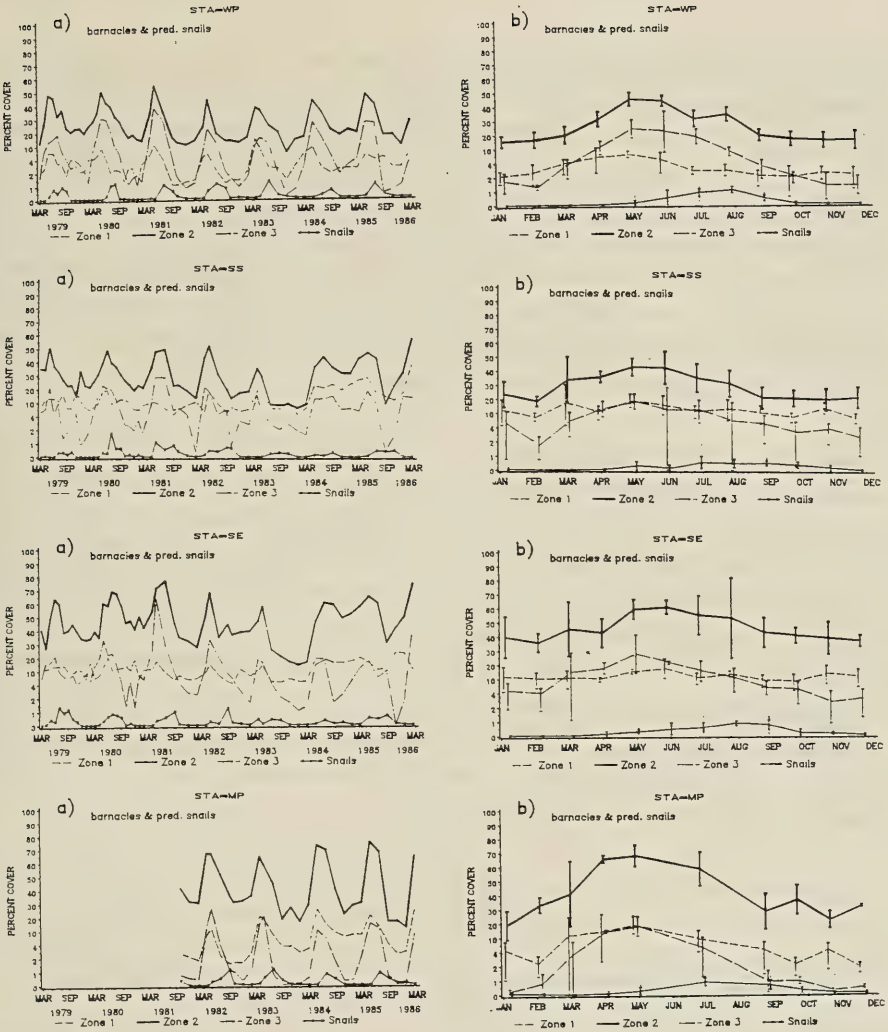

Figure 6. (cont.) Abundance of barnacles and predatory snails as percent cover in each zone: (a) from March 1979-March 1986, and (b) monthly from 1979-1986, for WP, SS, SE, and MP.

and coverage increases through early summer as individuals grow. By late summer, barnacle coverage decreases, as individuals are lost to predation (especially in Zone 3), or desiccation (especially in Zone 1). Barnacles may settle so densely or grow so quickly that they may eventually be lost to intraspecific competition. Crowding and 'hummocking' (cf. Grant 1977) weakens an individual's attachment to substratum, and clumps of barnacles may be lost, especially during autumn storms (Foertch and Keser 1981). Algae that have settled and grown on or between barnacles may also be lost with the removal of barnacles. Another factor contributing to local barnacle mortality is predation; the most important predators in the Millstone Point area are two carnivorous snails, *Urosalpinx cinerea* (oyster drill) and *Thais lapillus* (dog whelk). These snails are most abundant in late summer, and are partially responsible for the decline in barnacle abundance (cf. Hanks 1957; Bayne and Scullard 1978; Foertch and Keser 1981).

Barnacles are also lost as a consequence of thermal stress. Throughout the NUEL rocky shore sampling program, thermal stress was a source of mortality only at Fox Island-Exposed, and only after the opening of a second quarry cut in August 1983. Barnacle mortality was one of the changes observed in the rocky shore community near the MNPS discharge following the opening of the second cut; the changes are described and discussed in Appendix IV. Briefly, changes in water circulation patterns allowed incursion of warm water to areas that were previously unimpacted (Fox Island-Exposed and the shoreline between FE and the MNPS discharges). In spring, when ambient water temperatures were below 5 °C, the thermal incursion was not detrimental to barnacles; at the time of peak barnacle settlement, maximum temperatures at FE were ca. 12-14 °C. In March and April, dense barnacle set and rapid growth were observed. However, in autumn, water temperature at FE exceeded 28 °C (see earlier Temperature Data section), and barnacles were eliminated from this site.

Fucus vesiculosus is another important component in the northern Atlantic and local rocky shore communities (Fig. 7). Peak abundance during the year usually occurs in late summer, following growth of germlings that appear in spring; substratum coverage declines as plants are lost to autumn and winter storms. There is also station-to-station variability in *Fucus* abundance; some stations have very low *Fucus* cover (e.g., BP), others have consistently high cover (e.g., WP, SE). These patterns are related to degree of exposure to waves; moderately exposed stations favor *Fucus* populations (Topinka et al. 1981). *Fucus* cover also varies from year to year. At some stations (e.g., GN, FS), there appears to be a 3-5 year cycle of *Fucus* abundance, related to the ecological life-span of this alga. *Fucus* does not propagate vegetatively from a basal holdfast; rather, it occupies new substrata following settlement of zygotes and growth of

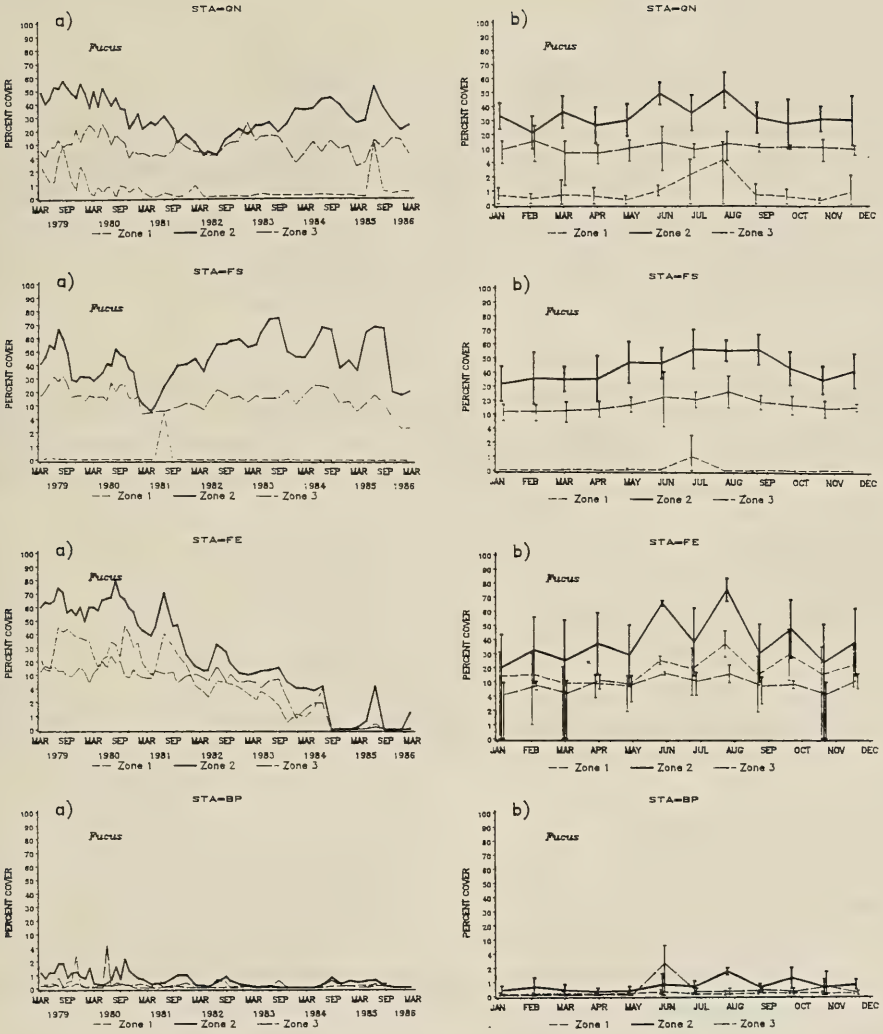
STA=GN
a)
Fucus
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 1986
Zone 1 Zone 2 Zone 3
STA=GN
b)
Fucus
PERCENT COVER
JAN FEB MAR APR MAY JUN JUL AUG SEP OCT NOV DEC
Zone 1 Zone 2 Zone 3
STA=FS
a)
Fucus
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 1986
Zone 1 Zone 2 Zone 3
STA=FS
b)
Fucus
PERCENT COVER
JAN FEB MAR APR MAY JUN JUL AUG SEP OCT NOV DEC
Zone 1 Zone 2 Zone 3
STA=FE
a)
Fucus
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 1986
Zone 1 Zone 2 Zone 3
STA=FE
b)
Fucus
PERCENT COVER
JAN FEB MAR APR MAY JUN JUL AUG SEP OCT NOV DEC
Zone 1 Zone 2 Zone 3
STA=BP
a)
Fucus
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 1986
Zone 1 Zone 2 Zone 3
STA=BP
b)
Fucus
PERCENT COVER
JAN FEB MAR APR MAY JUN JUL AUG SEP OCT NOV DEC
Zone 1 Zone 2 Zone 3

Figure 7. Abundance of *Fucus* as percent cover in each zone: (a) from March 1979-March 1986, and (b) monthly from 1979-1986, for GN, FS, FE, and BP.

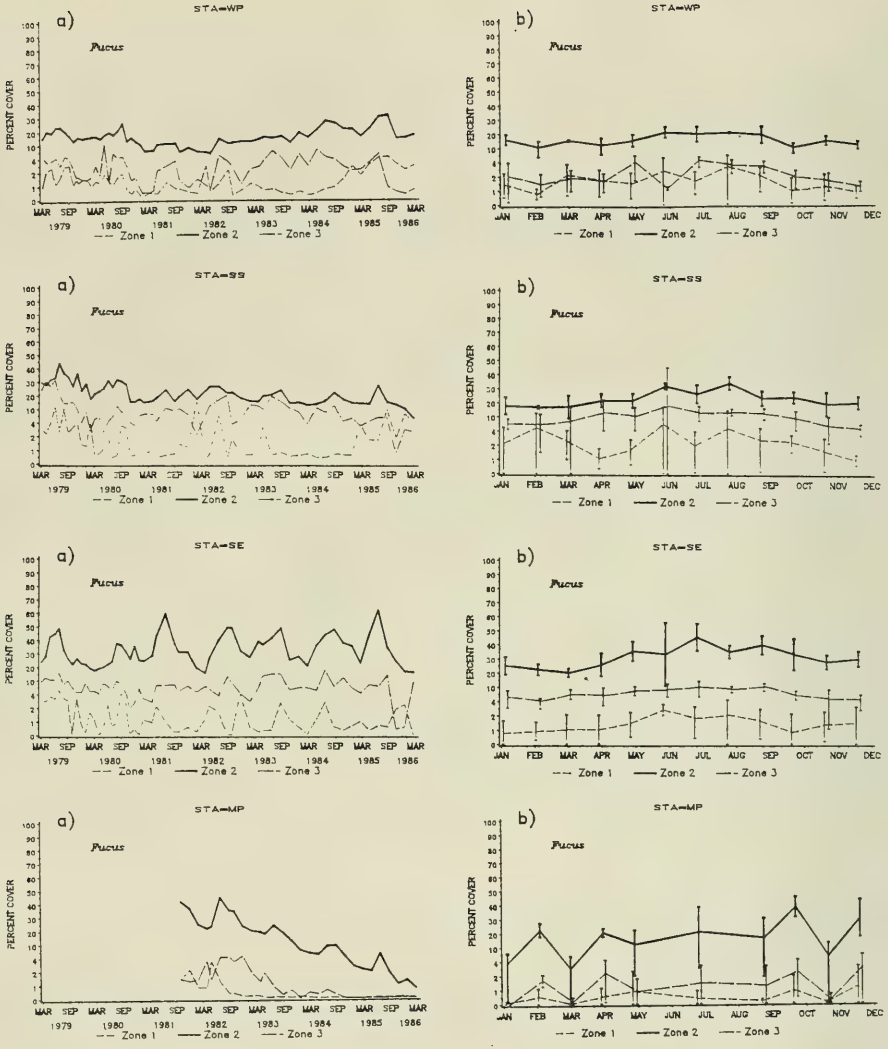

Figure 7. (cont.) Abundance of *Fucus* as percent cover in each zone: (a) from March 1979-March 1986, and (b) monthly from 1979-1986, for WP, SS, SE, and MP.

germlings (Knight and Parke 1950; Keser and Larson 1984). Typically, these germlings do not grow under an established *Fucus* canopy. However, if an area in the mid intertidal is cleared (e.g., by ice-scour), *Fucus* zygotes settle and grow into a new canopy, composed of plants of similar age. As these plants mature, they become increasingly susceptible to epiphytism (Menge 1975), storm damage and ice-scouring (Mathieson et al. 1982; Chock and Mathieson 1983). These processes tend to remove many plants at once; plant loss opens new substrata for colonization, and the cycle of *Fucus* abundance is maintained (cf. Schonbeck and Norton 1980; Keser and Larson 1984).

The decrease in *Fucus* cover at FE from 1980 through 1983 was another example of a cyclic pattern in abundance. However, thermal impact resulting from water temperatures in excess of 28 °C interrupted the *Fucus* population cycle at FE. The increase predicted after a settlement of germlings in spring 1984 failed to occur after thermal incursion and subsequent germling mortality.

Fucus cover also decreased at Millstone Point, the station second closest to the MNPS discharges. This decline may represent the descending portion of the same type of abundance cycle seen at FE, or it may reflect a direct or indirect thermal effect. A direct effect might relate to the thermal tolerance of *Fucus*; an indirect effect might be related to the observed increase in abundance of *Littorina littorea* at MP, and concomitant increase in grazing pressure on newly settled *Fucus* germlings. Other researchers have shown that high grazer densities can retard *Fucus* recolonization for several years (Lubchenco 1983; Keser and Larson 1984).

Chondrus crispus is the dominant alga in low intertidal (Zone 3) and shallow subtidal areas near MNPS, as it is throughout New England, the Canadian Maritimes, and Northern Europe (Mathieson and Prince 1973). *Chondrus* populations were stable at all NUEL study sites, both within and among years (Fig. 8). The decline in abundance seen at FE after the opening of the second quarry cut was due to water temperatures in excess of 28 °C which eliminated *Chondrus* from FE in September 1984.

Typically, *Chondrus* is host to a variety of ephemeral epiphytic algae. In our area, the dominant epiphytes on *Chondrus* are *Monostroma pulchrum*, *Polysiphonia harveyi*, and *P. novae-angliae*. These algae have distinct seasonal peaks of abundance (Fig. 8), with *Monostroma* occurring in spring and *Polysiphonia* in late summer-autumn. Some researchers have reported that shading by epiphytes is harmful to the

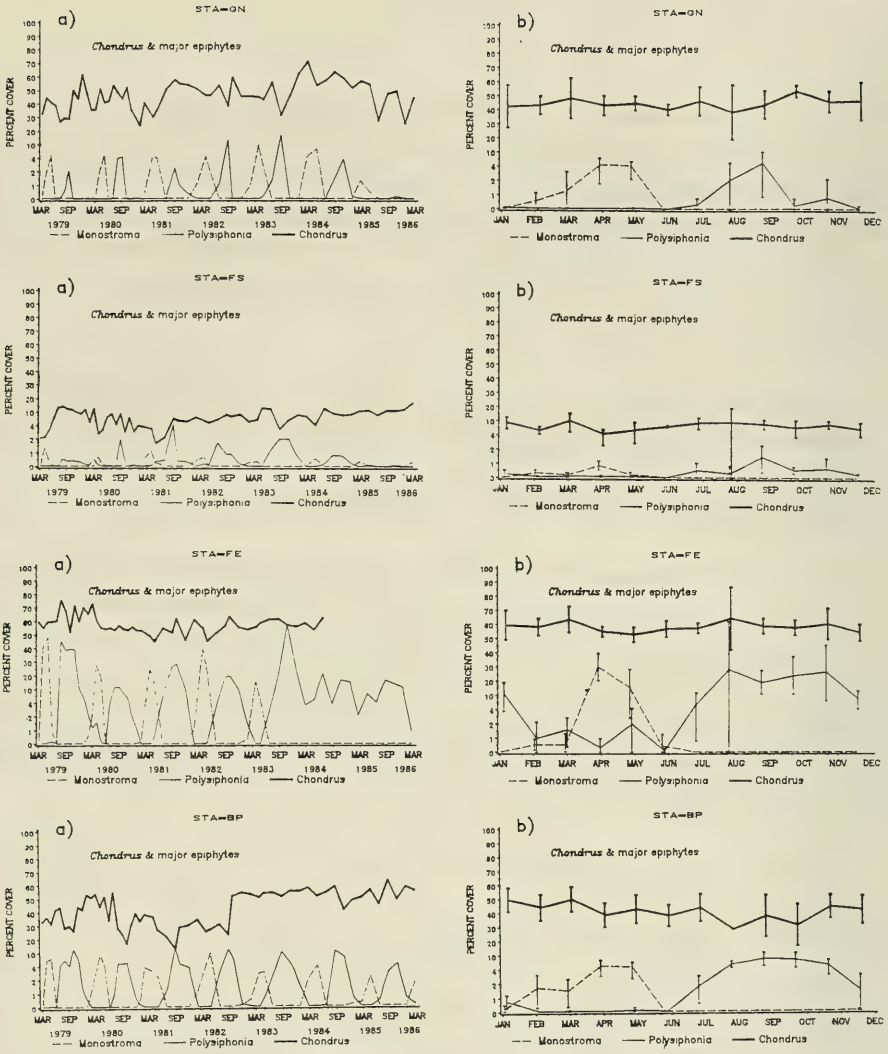

Figure 8. Abundance of *Chondrus* and major epiphytes as percent cover in each zone: (a) from March 1979-March 1986, and (b) monthly from 1979-1986, for GN, FS, FE, and BP.

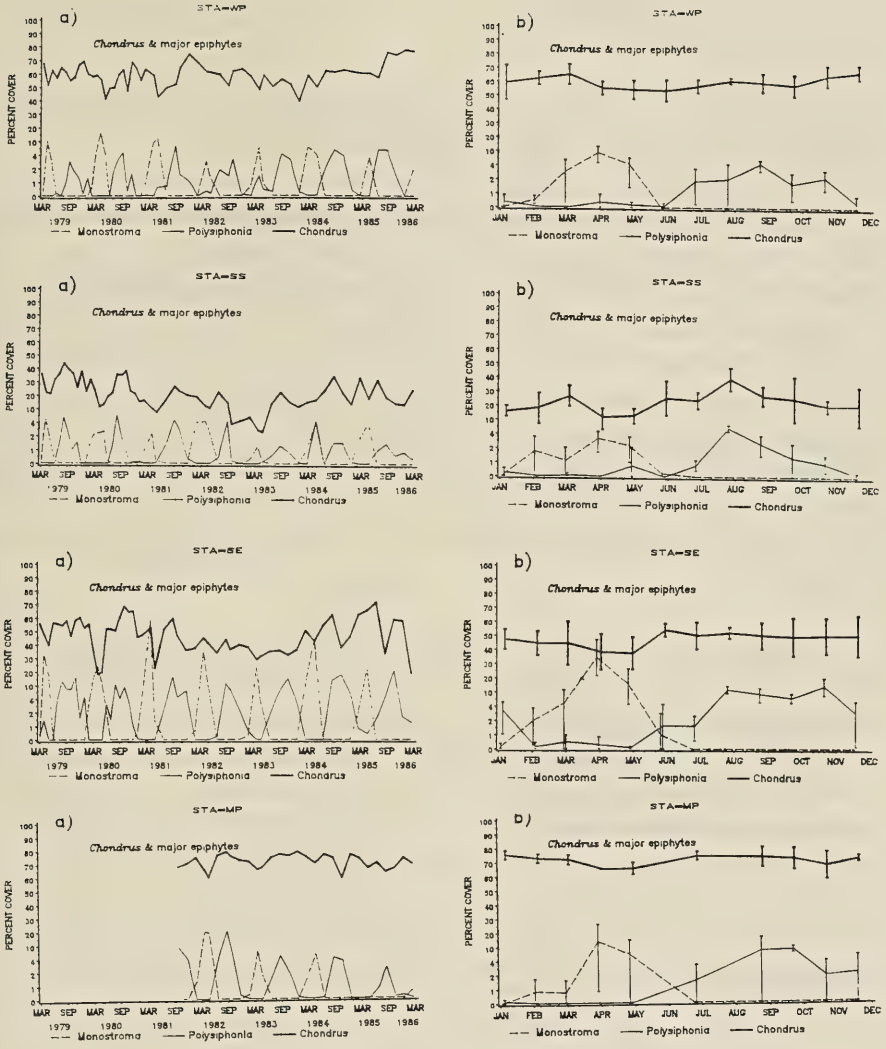

Figure 8. (cont.) Abundance of *Chondrus* and major epiphytes as percent cover in each zone: (a) from March 1979-March 1986, and (b) monthly from 1979-1986, for WP, SS, SE, and MP.

underlying *Chondrus* (Menge 1975; Lubchenco and Menge 1978); no such detrimental relationship was evident from our data. *Chondrus* maintained high understory abundance.

Many other species of ephemeral algae contribute to the local intertidal community. Grouped by division, some patterns are evident (Figs. 9 and 10). Generally, ephemeral algae are more common in Zone 3 than in Zone 2, and more common at exposed than at sheltered stations. Many of these species are opportunistic, and may occur at any time of year (e.g., *Ulva lactuca*, *Enteromorpha* spp.), but others show seasonal periodicity (cf. Qualitative Studies section). For example, an increase in red algal abundance in early spring was attributed to an increase of *Bangia* and *Porphyra* spp. Similarly, ephemeral browns were most abundant in early summer (Apr.-July), corresponding to peak abundance of *Petalonia* and *Scytosiphon*. Green algae, especially *Enteromorpha* spp., were most common in early spring and autumn. Some of these abundance patterns were related to numbers and activity of grazers (Figs. 9 and 10). In the MNPS area, littorinid snails (especially *Littorina littorea*) are the most important grazers, most active in summer.

The abundances of major components of local rocky shore communities, and therefore the structure and appearance of the communities themselves, vary over time and space. Variations are predictable, and explainable in terms of seasonality, degree of exposure, and life-history of the organisms. Except for the alteration noted at FE following the opening of the second quarry cut, changes to local rocky shore communities have been minor, indicating stable environmental conditions during 2-unit operation. This stability permits a characterization of each rocky shore station in terms of patterns of abundance of its major community components. A measure of this stability is illustrated in Figure 11. Each station/year combination is represented by the average annual percent cover of each species; Bray-Curtis similarity indices were calculated for each pair-wise comparison. When these comparisons are plotted by applying a clustering algorithm to the similarity matrix, the resultant dendrogram shows several levels of grouping.

The first level of separation distinguishes between unimpacted (group I) and impacted (group II) collections. As previously described, the community that developed at FE in response to elevated water temperatures was more like that found in the effluent quarry than at other sampling sites. Detection of community alterations are evident from analyses of this type.

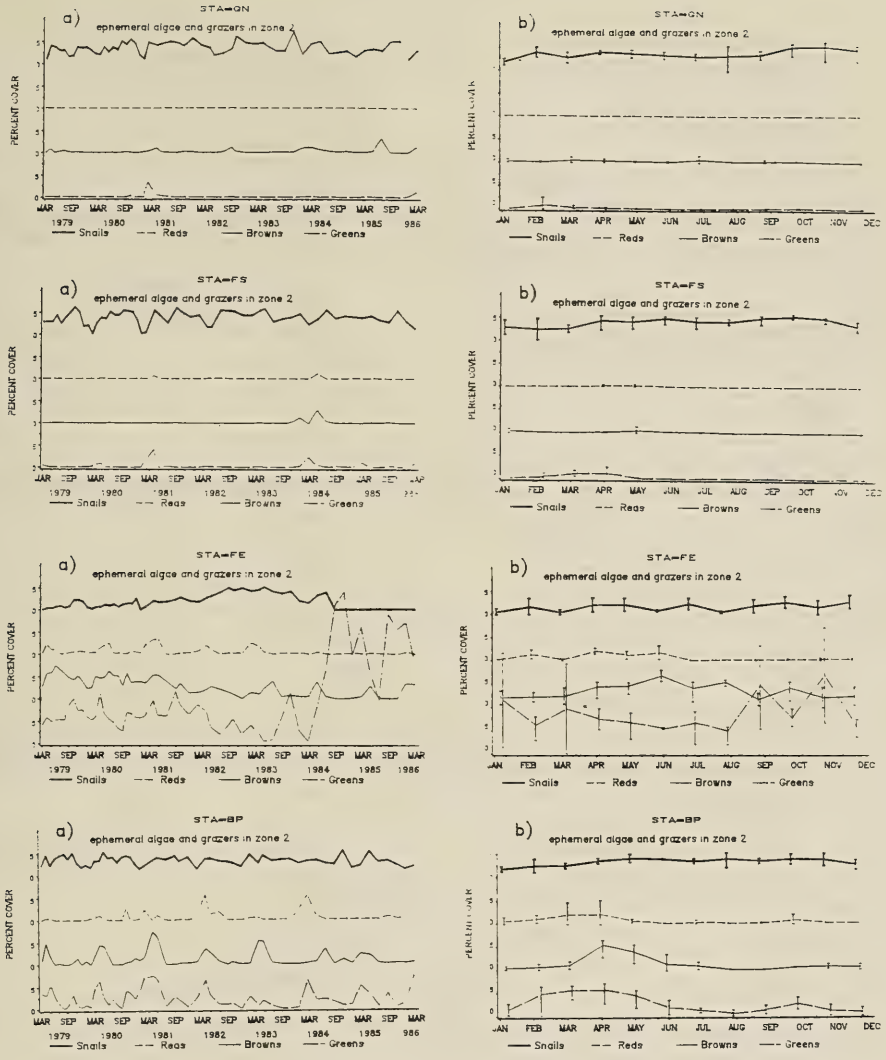
STA=GN
a)
ephemeral algae and grazers in zone 2
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 986
Snails Reds Browns Greens
b)
ephemeral algae and grazers in zone 2
JAN FEB MAR APR MAY JUN JUL AUG SEP OCT NOV DEC
STA=FS
STA=FE
STA=BP

Figure 9. Abundance of ephemeral algae and grazers in Zone 2 as percent cover: (a) from March 1979-March 1986, and (b) monthly from 1979-1986, for GN, FS, FE, and BP.

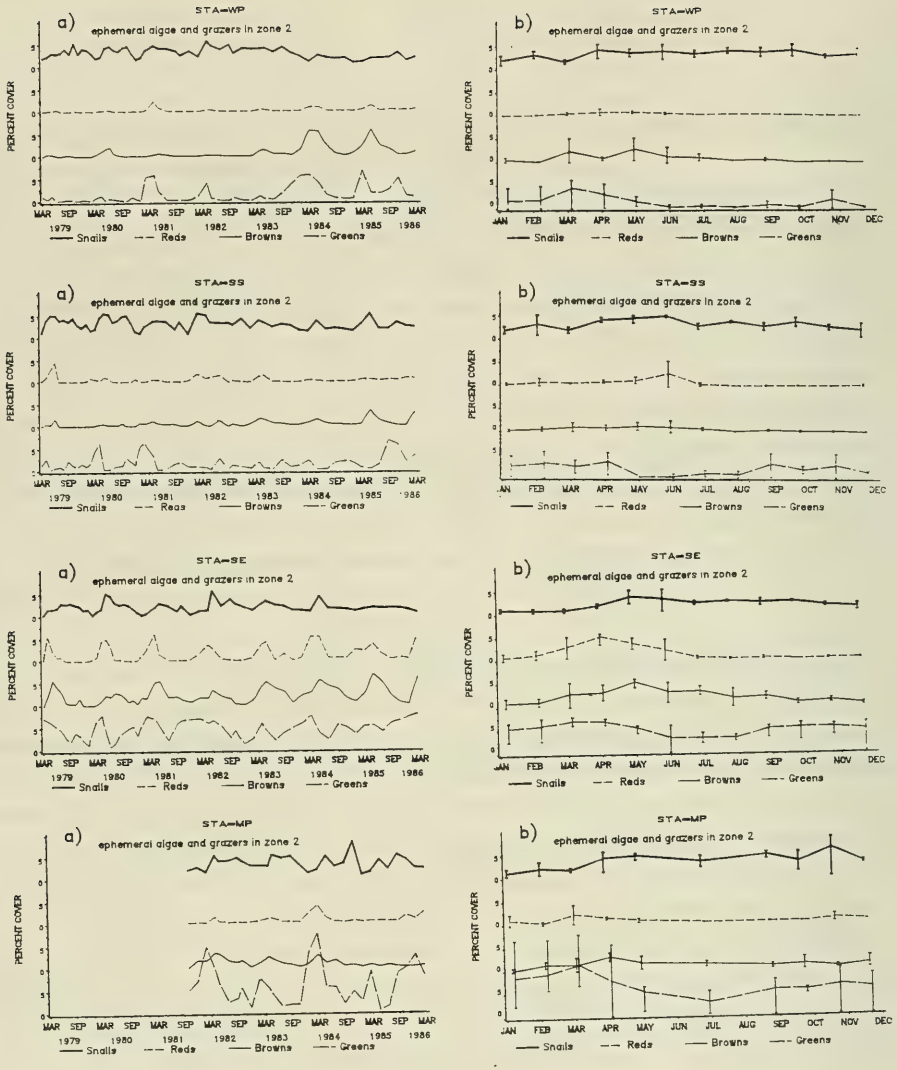

Figure 9. (cont.) Abundance of ephemeral algae and grazers in Zone 2 as percent cover: (a) from March 1979-March 1986, and (b) monthly from 1979-1986, for WP, SS, SE, and MP.

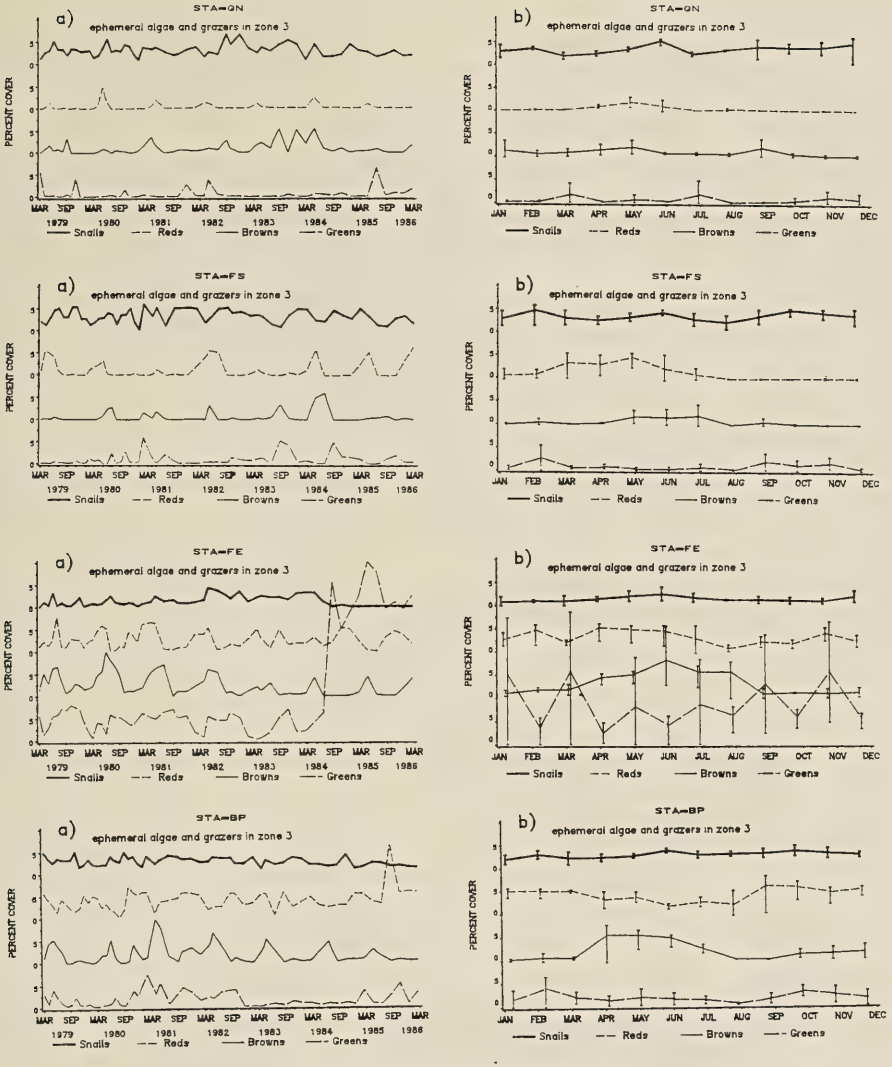

Figure 10. Abundance of ephemeral algae and grazers in Zone 3 as percent cover: (a) from March 1979-March 1986, and (b) monthly from 1979-1986, for GN, FS, FE, and BP.

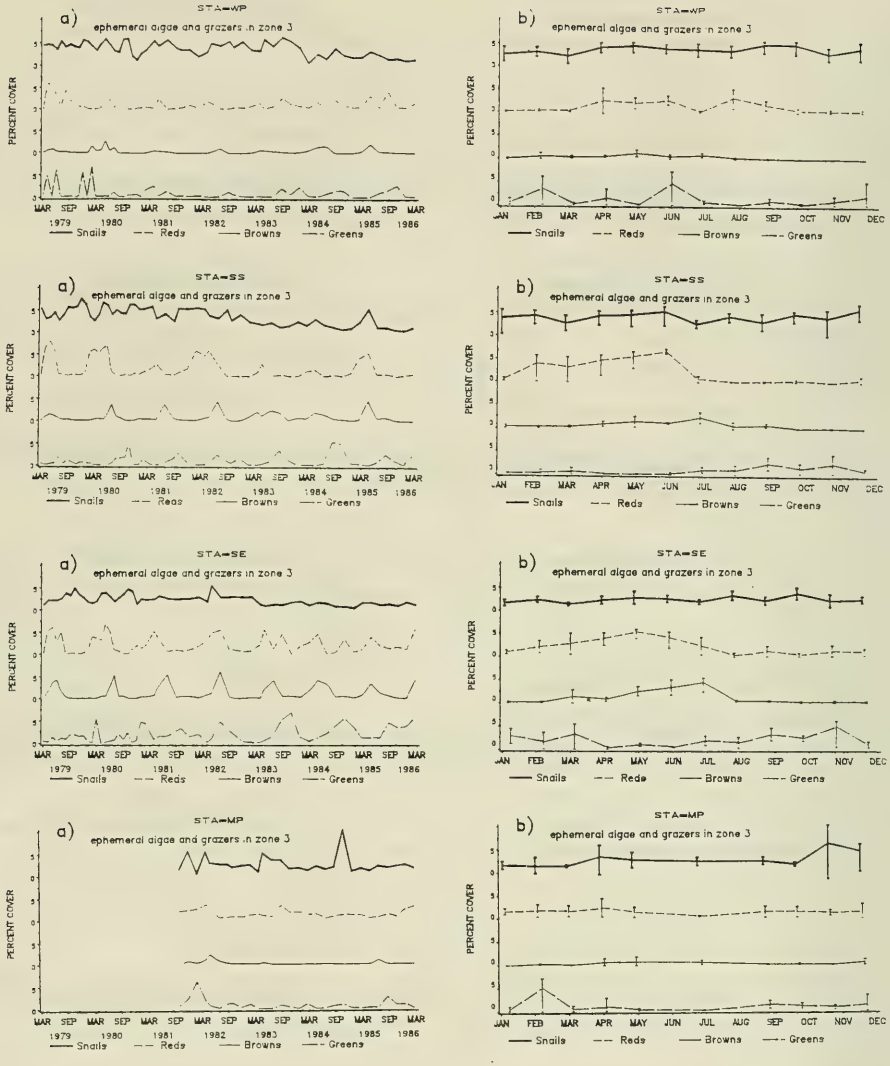

Figure 10. (cont.) Abundance of ephemeral algae and grazers in Zone 3 as percent cover: (a) from March 1979-March 1986, and (b) monthly from 1979-1986, for WP, SS, SE, and MP.

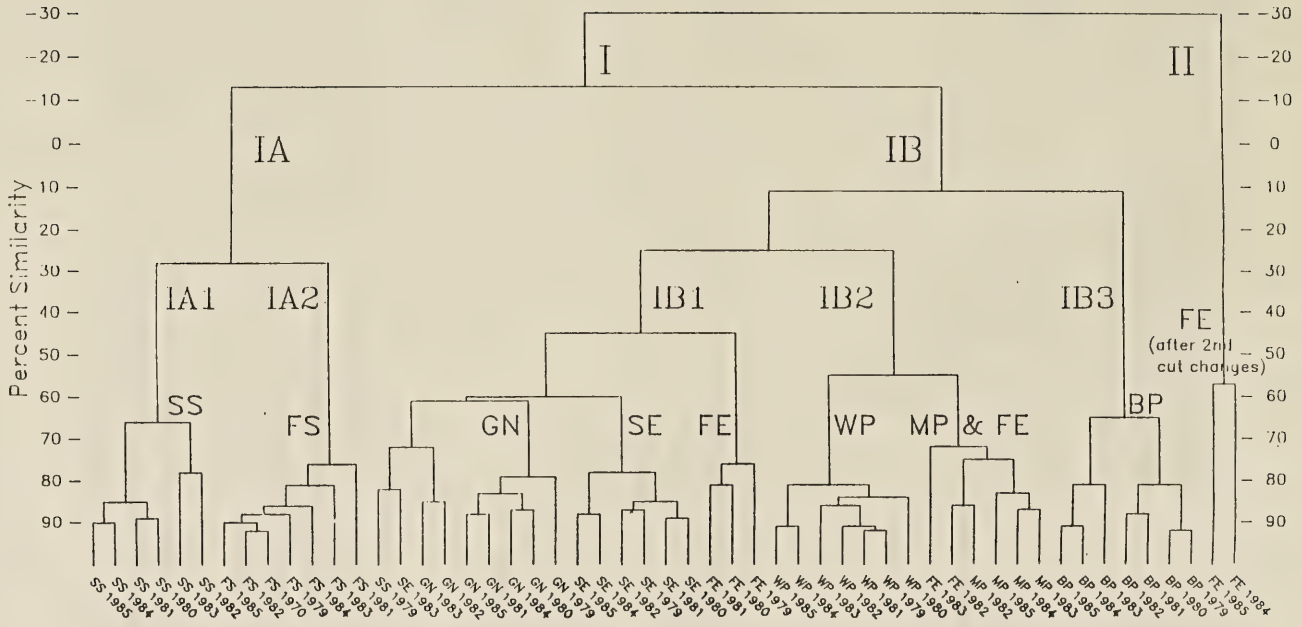

Figure 11. Clustering dendrogram of percent similarity, by station and year, 1979-1985.

Unimpacted stations may be separated on the basis of those with little *Chondrus* cover in Zone 3 (< 25% on average; group IA) and those with greater *Chondrus* cover (> 40%; group IB). These groupings may be further subdivided on the basis of *Fucus* cover; groups IA2 (FS) and IB1 average 30-40% *Fucus*, groups IA1 (SS) and IB2 average about 10%, and group IB3 (BP) with only a trace of *Fucus*.

Further subdivisions are based on species complexes unique to each station, and consistent over time. The obvious pattern is for each station to resemble itself more than any other station. Fox Island-Exposed is the only station to show clear changes over time. Collections from FE separate into periods when *Fucus* was abundant (1979-81), when *Fucus* was less abundant (and collections were similar to those at Millstone Point, 1982-83), and the period when FE was thermally impacted (1984-85).

In summary, local rocky intertidal communities, as represented in the undisturbed transects, have undergone drastic changes in the immediate vicinity of MNPS, little change elsewhere. Analyses of data collected during 2-unit operation show that community parameters vary within predictable limits, and that the observed patterns of distribution and abundance of intertidal organisms may be explained in terms of, e.g., seasonality, degree of exposure, intertidal height, inter- and intraspecific competition. Similar analyses, using data collected during 3-unit operation, coupled with analyses of individual community components, permit assessment of possible changes to nearby rocky shore communities, especially those that may be affected by the 3-Unit thermal plume.

Recolonization Studies

Transects

Recolonization of denuded areas on local rocky shores is an on-going, naturally occurring process; grazing/predation, storms, senescence, ice/sand scour all clear areas of intertidal rock, and make space available for recolonization. Our recolonization experiments, therefore, allow isolation and identification of some factors that influence the structure of local rocky shore communities. Juvenile stages of recolonization organisms in the denuded transects could be more susceptible to power-plant effects than adults in established populations, and comparisons of rates and patterns of community recovery under the exclusion cages permit determination of the influence of grazing and predation on local intertidal community structure.

As shown in the previous section (Undisturbed Transects), the shore community is divided into three zones, each dominated by a specific biota: barnacles in the high intertidal, barnacles and fucoids in the mid intertidal, and *Chondrus* in the low intertidal. Because of different life history strategies, each component recovers at a different rate. Other factors that affect rates of recolonization include degree of exposure, season of denuding, and inter- and intraspecific competition (Keser and Larson 1984). For example, the high intertidal is mostly barren rock, on which ephemeral algae, barnacles, and snails appear seasonally. Many of the ephemeral algae are opportunistic, and readily colonize (at least temporarily) available space. Mobile intertidal predators repopulate cleared substrata from nearby undisturbed areas as soon as food availability and weather conditions permit. Therefore, the high intertidal may appear 'recovered' immediately after denuding, and similar in appearance to Zone 1 of nearby undisturbed transects at the end of the first barnacle set. Percent cover values for all taxa found in the recolonization transect studies are presented in Appendix RS Ic.

The mid intertidal of the local rocky shore community is dominated by a fucoid canopy over a barnacle understory (Figs. 6 and 7). Recovery of these components is illustrated by plotting their abundance (as percent cover) over time, compared with abundance in the undisturbed transects (controls). Figure 12 represents recolonization by barnacles in Zone 2 of the recolonization transects. Following the spring 1979 denuding (made prior to the peak barnacle settlement period), barnacles settled heavily and, generally, within two months were at least as abundant in the recolonization strips as in the control areas. The exception was Giants Neck; at this station, most barnacles had already set at the time of denuding, and subsequent settlement was lighter than at other stations. These barnacles grew through summer, and by October 1979 barnacle coverage in the recolonization strips was as high as that in the controls. After the autumn 1981 denuding (after barnacle set), barnacles were rare or absent in recolonization strips at all stations until the following spring, after which their abundance paralleled that in the controls. The same seasonal patterns of abundance were seen, i.e., settlement in spring, growth into summer, then decline in cover in autumn and winter (cf. Undisturbed Transects section). At FE, patterns of barnacle abundance after the opening of the second cut (specifically, complete elimination in September 1984 and September 1985) were related to thermal incursion, and are included as part of the community changes described and discussed in Appendix RS IV.

Recovery of the *Fucus* population in Zone 2 of the recolonization strips, relative to that in the undisturbed transects (Fig. 13), was directly related to degree of exposure, hence the amount and duration

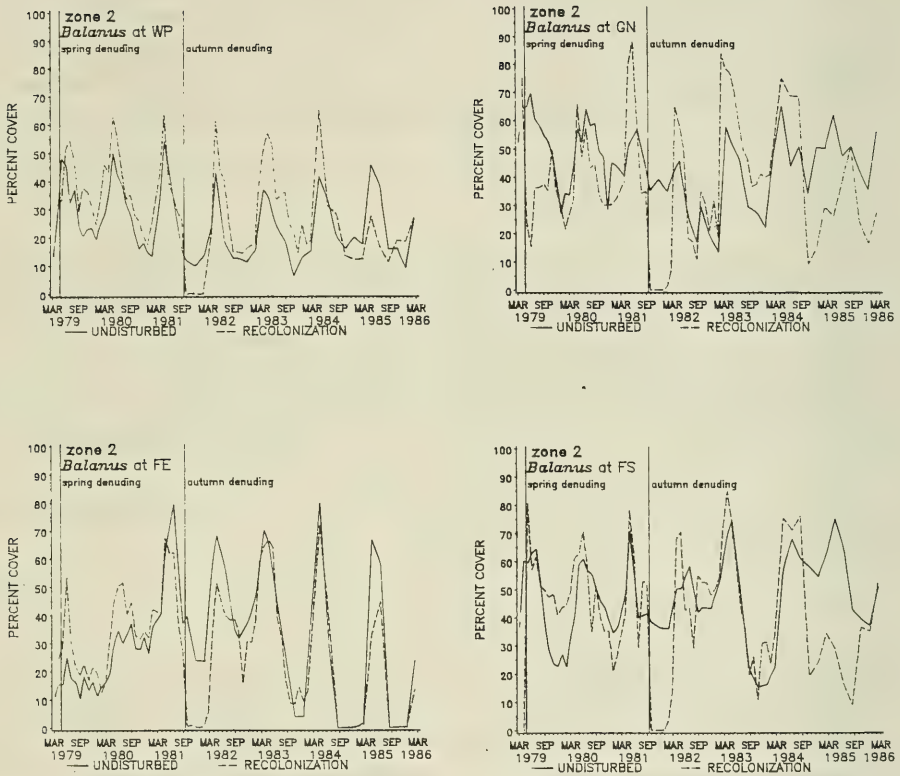
zone 2
Balanus at WP
spring denuding
autumn denuding
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 1986
UNDISTURBED
RECOLONIZATION
zone 2
Balanus at GN
spring denuding
autumn denuding
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 1986
UNDISTURBED
RECOLONIZATION
zone 2
Balanus at FE
spring denuding
autumn denuding
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 1986
UNDISTURBED
RECOLONIZATION
zone 2
Balanus at FS
spring denuding
autumn denuding
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 1986
UNDISTURBED
RECOLONIZATION

Figure 12. Abundance of *Balanus* at recolonization sites in Zone 2, from March 1979-March 1986. Vertical lines represent time of denuding.

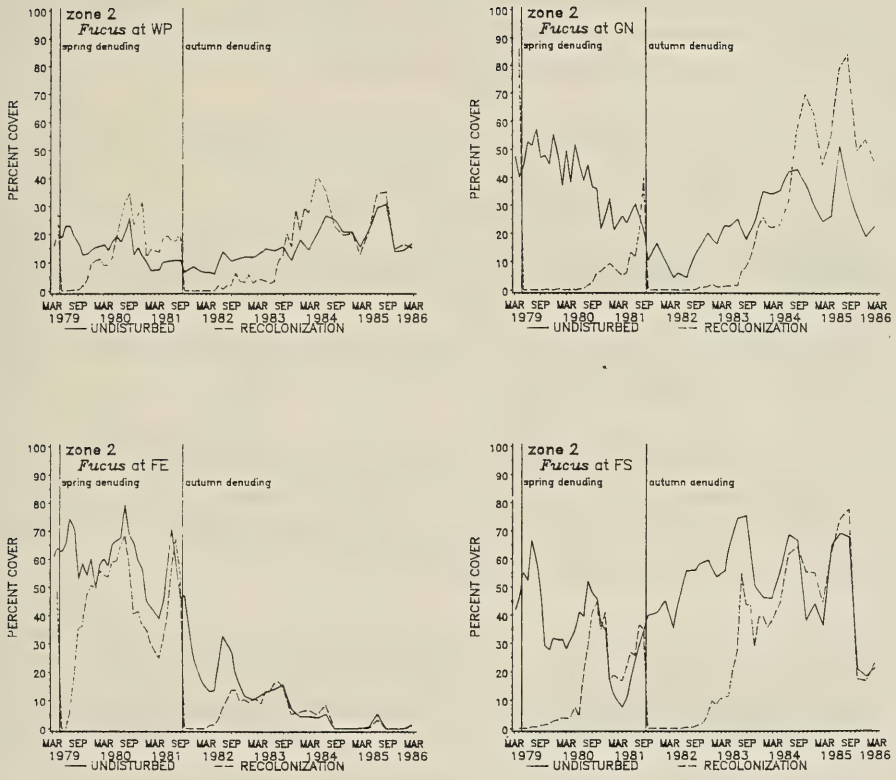

Figure 13. Abundance of *Fucus* at recolonization sites in Zone 2 from March 1979-March 1986. Vertical lines represent time of denuding.

of available moisture. Following spring denuding, *Fucus* recovered at FE (the most exposed recolonization station) in 8 months, at WP (2nd most exposed) in 14 months, at FS (2nd most sheltered) in 19 months, and at GN (the most sheltered) in 29 months. After autumn denuding, the time needed for *Fucus* abundance in the recolonization strips to approach that in the controls at FE, WP, FS, and GN were, respectively, 16, 24, 36, and 36 months. Delay was attributed to the requirement for surface heterogeneity to provide refuges from grazing; *Littorina littorea* prevent *Fucus* from establishing itself on smooth surfaces (Lubchenco 1983). The community of the mid intertidal recovers (in terms of *Fucus* canopy) in as little as 8 months (when denudation occurs prior to barnacle set, at an exposed station), or as long as 3 years (when denudation occurs after barnacle set, at a sheltered station).

Recovery of the low intertidal (re-establishment of a *Chondrus* canopy) takes longer than recovery of high or mid intertidal areas (Fig. 14). *Chondrus* propagates vegetatively from a basal crust; if the upright axes are removed (e.g., by scraping or freezing) but the crust left intact, recovery of uprights may be rapid (MacFarlane 1956). If, however, the crust is removed (e.g., by our methods of scraping and burning), repopulation must occur by settlement of spores. For *Chondrus*, repopulation initiated by spores is a slow process. After both spring and autumn denudings, *Chondrus* at GN and WP reached a maximum of only 5% cover after 30 months. In fact, during each 30 month experiment, the highest abundance of *Chondrus* at any recolonization station was ca. 10% at FS after the autumn denuding.

Recovery of a *Chondrus* population involves long-term survival of relatively slow growing individuals. This strategy is at the opposite end of a spectrum from that of ephemeral algae, that settle and grow quickly whenever conditions are favorable. It also implies that even short-term periodic exposure to lethal conditions will preclude re-establishment of *Chondrus* at FE.

Even under favorable conditions, interspecific competition for space from *Fucus vesiculosus* may partially explain the slow recovery rate of *Chondrus*. Regardless of the time of year in which denuding occurred, *Fucus* was the first perennial macroalga to colonize the low intertidal, and usually developed into a dense canopy (Fig. 15). Following experimental denudation of areas in dense *Chondrus* beds, Lubchenco (1980) reported that *Fucus* colonized and persisted for at least 3 years. *Chondrus* settled and grew under the *Fucus* canopy. Lubchenco predicted that when *Fucus* senesced (after its 3-5 year lifespan), *Chondrus* would remain and exclude further *Fucus* settlement. In other words, *Fucus* initially out-competes *Chondrus*, but eventually *Chondrus* would dominate the low intertidal. Data from recolonization transects

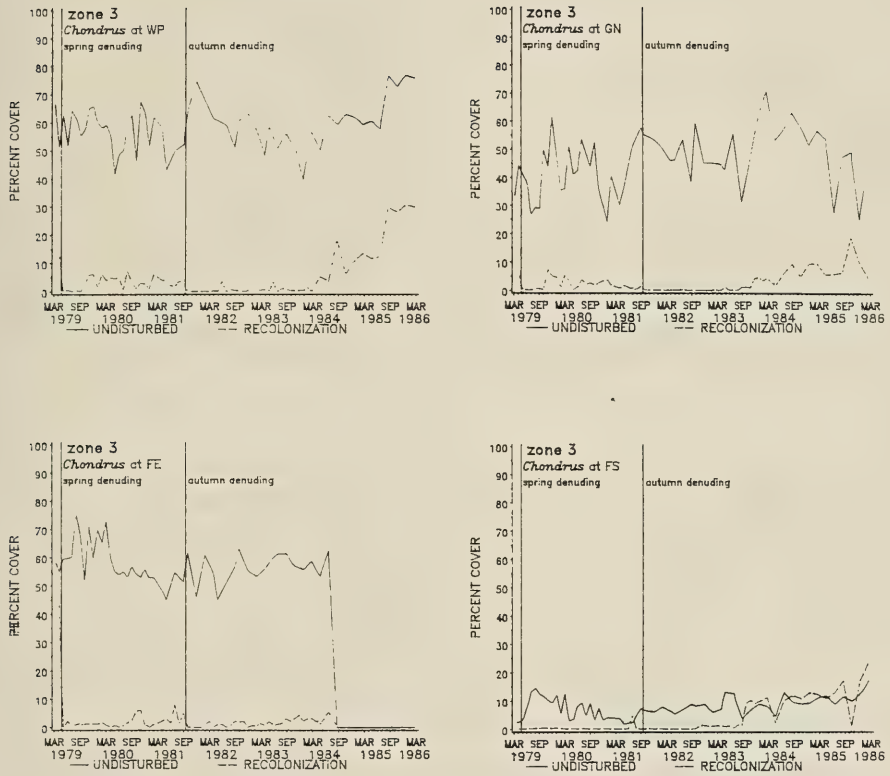

Figure 14. Abundance of *Chondrus* at recolonization sites in Zone 3, from March 1979-March 1986. Vertical lines represent time of denuding.

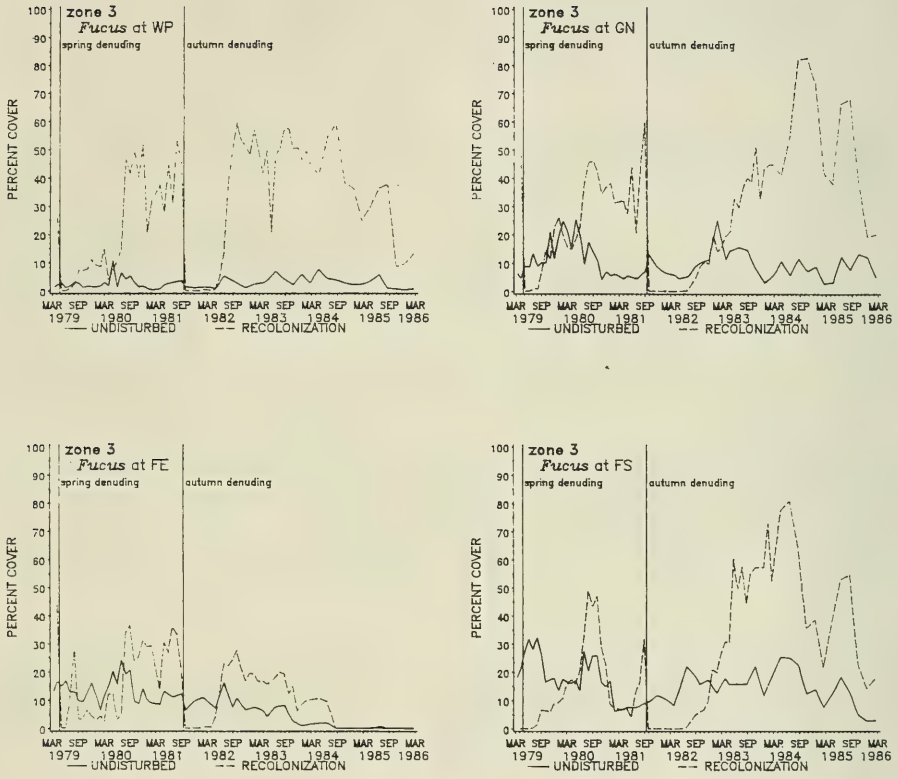

Figure 15. Abundance of *Fucus* at recolonization sites in Zone 3, from March 1979-March 1986. Vertical lines represent time of denuding.

support this hypothesis; i.e., five years after denuding, *Chondrus* was established under the *Fucus* canopy in Zone 3 at FS, GN, and WP (Fig. 14).

Barnacles also settle heavily in low intertidal areas (Fig. 16), more heavily under a *Fucus* canopy (recolonization strips) than under a *Chondrus* canopy (controls). However, predatory snails have a greater portion of the intertidal period to feed in the low intertidal, and barnacle mortality is higher than in Zone 2. Seasonal variability of barnacles is therefore more pronounced.

Results from recolonization transect studies conducted under 2-unit operating conditions support the conclusions that: 1) different components of the community recover at different rates, 2) recovery is faster at exposed stations than at sheltered stations, 3) when present, grazing snails can maintain smooth rock surfaces free from algal colonization, and 4) they are prevented from doing so indefinitely because of surface heterogeneity provided by barnacles. Obviously, spatial and temporal distribution patterns of grazers and predators are important influences on the structure of local intertidal communities. We examine influences of grazing and predation on recolonization in the next section.

Exclusion cages

Results from four series of exclusion cage experiments supplement conclusions relating to the effect of seasonality on rates and patterns of recolonization, and provide information on the effects of grazing and predation. Reduction of interspecific competition separates the physical and biological factors that influence distribution of intertidal organisms. Thus, it is possible to demonstrate whether rates and patterns of recolonization at Fox Island-Exposed were directly influenced by proximity to the MNPS discharge. Each series of experiments consisted of nine caged areas and nine control areas at each of four stations; the information is synthesized into a model of community development, outlining common sequences of events and alternate pathways. Complete percent cover data for each experimental area during each denuding are included in Appendix RS Id.

General patterns of community development are illustrated in Figure 17. In each case, the vertical axis represents relative, not absolute, abundance; absolute values varied among and within stations and zones, but trends were similar. In general, values were higher at exposed stations than at sheltered, and higher in mid and low intertidal areas than in Zone 1, also higher under the cages than in the control areas.

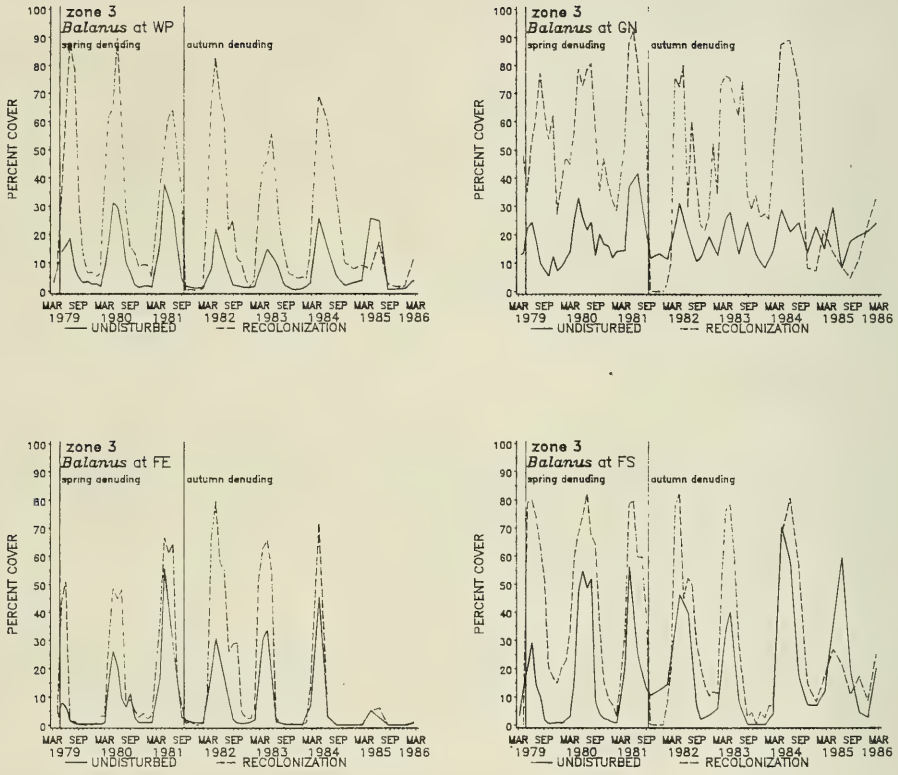
zone 3
Balanus at WP
spring denuding
autumn denuding
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 1986
UNDISTURBED
RECOLONIZATION
zone 3
Balanus at GN
spring denuding
autumn denuding
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 1986
UNDISTURBED
RECOLONIZATION
zone 3
Balanus at FE
spring denuding
autumn denuding
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 1986
UNDISTURBED
RECOLONIZATION
zone 3
Balanus at FS
spring denuding
autumn denuding
PERCENT COVER
MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR SEP MAR
1979 1980 1981 1982 1983 1984 1985 1986
UNDISTURBED
RECOLONIZATION

Figure 16. Abundance of *Balanus* at recolonization sites in Zone 3, from March 1979-March 1986. Vertical lines represent time of denuding.

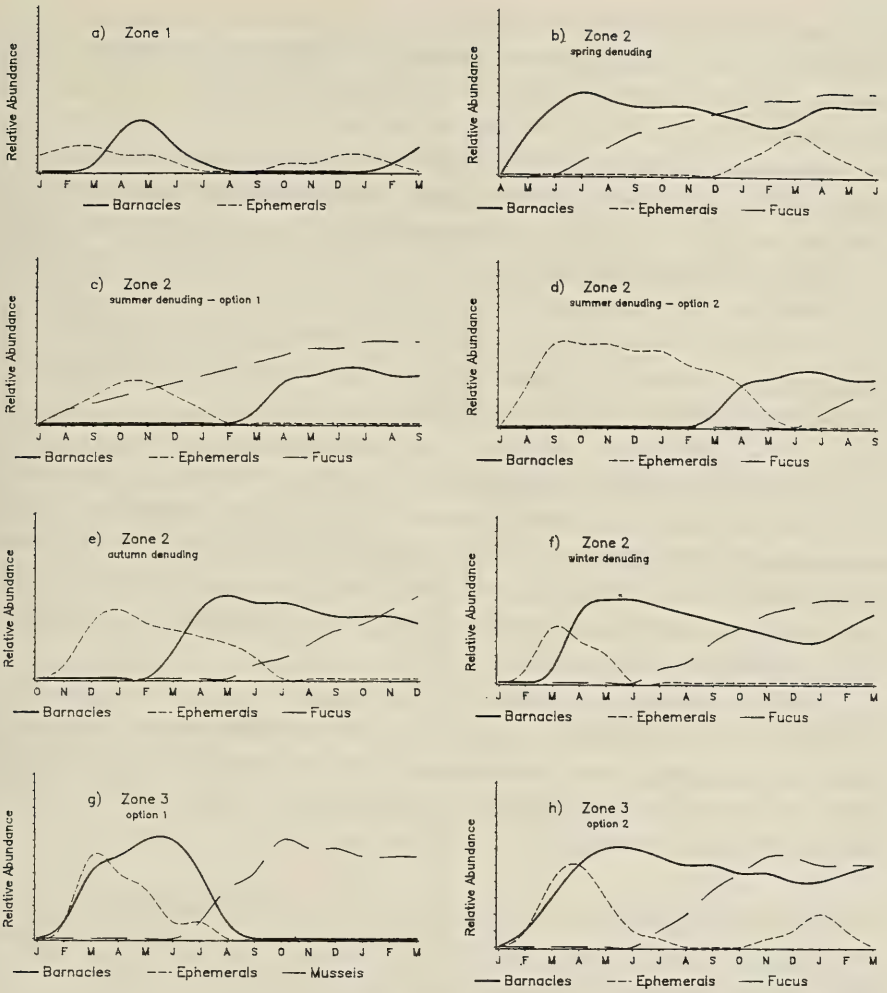

Figure 17. Models of recolonization community development under exclusion cages: a)Zone 1, b-f) different options in Zone 2, and g-h) different options in Zone 3.

As described in the previous section, the high intertidal (Zone 1) is mostly barren rock, with seasonal occurrences of barnacles and ephemeral algae. Herbivory contributes to the structure of high intertidal communities (Foertch and Keser 1981; Cubit 1984), but in general, physical factors are the controlling influence (Menge 1975). Thus, the ephemeral algal turf that appears in spring in Zone 1 cages and control areas is removed from control areas by grazing in early summer, and from caged areas by desiccation in late summer. Barnacle settlement and growth are higher under the cages than in controls, but similarly barnacles are usually lost from both areas with autumn desiccation. Degree of exposure had more effect on community structure than did protection from grazing and predation. Growth and survival of barnacles and ephemeral algae were higher at exposed stations (with more available moisture) than at sheltered stations. For example, maximum barnacle coverage in Zone 1 cages at an exposed station exceeds 90%; at a sheltered station less than 10% coverage. The seasonal cycles of settlement, growth, and mortality were independent of the time of year in which denuding occurred (Fig. 17a).

In mid intertidal cages, initial stages of recolonization were dependent mostly on what reproductive units were in the water column when space was made available, i.e., barnacles in spring, *Fucus* in summer, ephemeral algae throughout the year (Fig. 17b-f). Unlike the recolonization transects, where a barnacle set was prerequisite for development of a *Fucus* canopy (crevices between barnacles provided refuge for *Fucus* germlings that were otherwise vulnerable to grazing), substrata denuded in summer was colonized quickly (< 1 month) by *Fucus*, if grazers were excluded.

Alternatively, exclusion of grazers retarded *Fucus* colonization, by permitting monopolization of substratum by ephemeral algae. In such instances, opportunistic algae occupied available rock, and *Fucus* settlement was delayed until the following year. Both patterns are illustrated in the plot representing a summer denuding (Fig. 17c and d).

In all Zone 2 denudings, it was apparent that exclusion of predators and grazers from mid intertidal areas had more effect on the rate of recolonization than on the ultimate structure of the recolonization community. Each experiment ran for 15 months, but there was a clear indication that, had they continued, the caged communities would have resembled nearby "undisturbed" communities, i.e., a *Fucus* canopy over a barnacle understory. These findings support the conclusion that recovery from disturbance is deterministic, even if transition states are variable (Paine 1984).

Recolonization was different in Zone 3 (low intertidal). The recolonization community developed under the cages differently from the one that occurred when predators and grazers were not excluded. The most common sequence of events is illustrated in Fig. 17g.

In Zone 3, *Mytilus edulis* set in early summer, even if barnacles already occupied available substrata. Mussels gradually out-competed barnacles (and all other organisms) for space, and by autumn, typically occupied 100% of the caged substratum. The mussels persisted as long as the cages remained in place. The cages were removed when crowding prevented further community development. Mussels were lost within 2-3 days, either washed away as a mass, or heavily preyed upon by *Urosalpinx*.

Alternatively, recolonization in Zone 3 could resemble that seen in Zone 2, i.e., development of a *Fucus* canopy over a barnacle understory (Fig. 17h). These exclusion cage studies did not continue long enough to determine the ultimate development of the low intertidal community. However, recolonization transect studies discussed earlier show that re-establishment of a *Chondrus* population usually takes more than 3-5 years. Exceptions occur and stochastic processes cannot be ignored. In one instance (specifically, site 3 at White Point after the autumn denuding), *Chondrus* appeared in the cage area within 4 months, and within a year, exceeded 85% cover. By contrast, *Chondrus* settlement never exceeded 5% in any of the other experimental denudings. Obviously, recolonization is subject to natural variability.

One important result of the exclusion cage studies is the conclusion that recolonization at Fox Island-Exposed was not affected by proximity to the MNPS discharge; the patterns of development seen at FE were similar to those at other stations. The series of exclusion cage experiments was completed before the shore community was altered following the opening of the second quarry cut. These data will be particularly useful for assessing potential effects of 3-unit operation.

The exclusion cage studies also show that grazers and predators exert their influence on local intertidal communities, especially in mid and low intertidal areas, by preventing monopolization of space by a single species (cf. Menge 1975; Menge 1978). Grazers and predators help to maintain local communities in a state of dynamic equilibrium, a balance between recruitment and growth, and senescence and removal. This balance has persisted throughout the MNPS area during 2-unit operation. Where the community balance has been disturbed (i.e., at FE following the opening of the second quarry cut), the community responds to the changes as an entirety, and also in terms of its constituent species populations.

Ascophyllum nodosum Studies

Growth

Since 1979, the rocky intertidal monitoring program has included studies of *Ascophyllum nodosum*, a large perennial alga that is abundant in the low and mid intertidal areas locally, as well as throughout New England, the Canadian Maritimes, and Northern Europe. *Ascophyllum* has been studied extensively throughout its range, and its vegetative and reproductive phenology is well documented (David 1943; Printz 1959; Baardseth 1970a; Sundene 1973; Mathieson et al. 1976; Wilce et al. 1978). *Ascophyllum* growth rate has been shown to be sensitive to water temperature changes, especially increases to ambient temperature (Vadas et al. 1976, 1978; Stromgren 1977; Wilce et al. 1978; Keser and Foertch 1982). Because of the alga's response to water temperature change and its mode of linear growth, this alga is an important biomonitoring tool in the rocky intertidal program. Details of growth and mortality of local *Ascophyllum* are summarized below.

From April 1979 through May 1983, *Ascophyllum* plants (tips) at Fox Island grew significantly longer than those at White Point or Giants Neck (Fig. 18a); data from the reference stations did not vary between themselves. In the year representative of this growth pattern, 1982-1983, *Ascophyllum* tips grew longer because of the 2-3 °C ΔT at FL under single quarry cut conditions (see Temperature Data section), showing a higher growth rate earlier in spring and an extended growing season in late autumn (Fig. 18b). The increased tip length at FL resulted from faster growth from April to July. Growth rate during the remainder of the year was similar to growth rates at the control stations. Annual growth was similar to growth recorded for *Ascophyllum* populations throughout its geographical range (cf. Vadas et al. 1976, 1978; Stromgren 1977, 1983; Wilce et al. 1978; Keser and Foertch 1982).

During the 1983-1984 growing season, FL water temperatures were 2-3 °C above ambient from April to July, and average tip length was significantly longer at FL than at the control sites (Fig. 19a). After August 1983 when the second quarry cut opened and only one unit was in operation, the water temperature at FL rose to 7-9 °C above ambient and plants were exposed to a maximum temperature of about 27 °C. Growth rate at FL decreased sharply from August to October when temperatures decreased (Fig. 19b); tissue damage and some deformed tips were observed. When both units were operating in spring 1984, elevated water temperatures (12-13 °C above ambient, ca. 20 °C at the end of April 1984) were within the

range of temperatures for optimal growth of *Ascophyllum*. Average tip length and growth rate were significantly greater at FL than at the control sites.

When two units were operating, with discharge through two quarry cuts, as in the 1984-1985 growing season, water temperature averaged 12-13 °C warmer than at the control sites, resulting in high initial growth at FL in April-June (Fig. 20a). In July, temperatures exceeded 25 °C and growth rate at FL decreased sharply (Fig. 20b); in July some tagged plants had tiny bladders and many lateral proliferations. By August, water temperatures exceeded 28 °C and *Ascophyllum* plants died at FL. *Ascophyllum* at GN and WP had a predictable epiphyte flora and appeared healthy. Other researchers have related increased water temperatures to physiological stress. Chock and Mathieson (1979) found the maximum net photosynthesis for summer *Ascophyllum nodosum* plants to occur at temperatures between 18-21 °C with a "conspicuous decrease" beyond 24 °C. Thermal injury to *Ascophyllum* was determined between 30-35 °C (Kanwisher 1966), while enhancement at 22 °C, gradual demise at 26 °C, and complete thallus destruction at temperatures above 30 °C was modelled by Vadas et al. (1978).

A second Fox Island station was established in April 1985, following the loss of *Ascophyllum* at FL in 1984. This new station was located at the first available *Ascophyllum* population around Fox Island-Exposed, approximately 200 m from the discharges (Fig. 2); water temperature at FN was 0-2 °C warmer than at controls in spring 1985 (with only two units in operation). *Ascophyllum* growth at FN was higher than at the control sites from April to May, but for the remainder of the growth season neither average tip length nor growth rate differed significantly between FN and GN (Fig. 21).

Temperature measurements made during periods when Unit 3 was testing its circulating water pumps in summer of 1985 (thereby increasing effluent volume) indicate that during 3-unit operation, FN will be exposed to water temperatures 2-3 °C above ambient. Thermal plume predictions indicate that WP may also be exposed to warmer water. Comparison of *Ascophyllum* growth under 3-unit operating conditions to data collected during 2-unit operation will permit assessment of the extent of the plume.

Mortality

Ascophyllum mortality, determined as thallus breakage, is a response to mechanical and environmental stress. Thallus breakage could occur above the base tag, either between the base tag and the colored tie

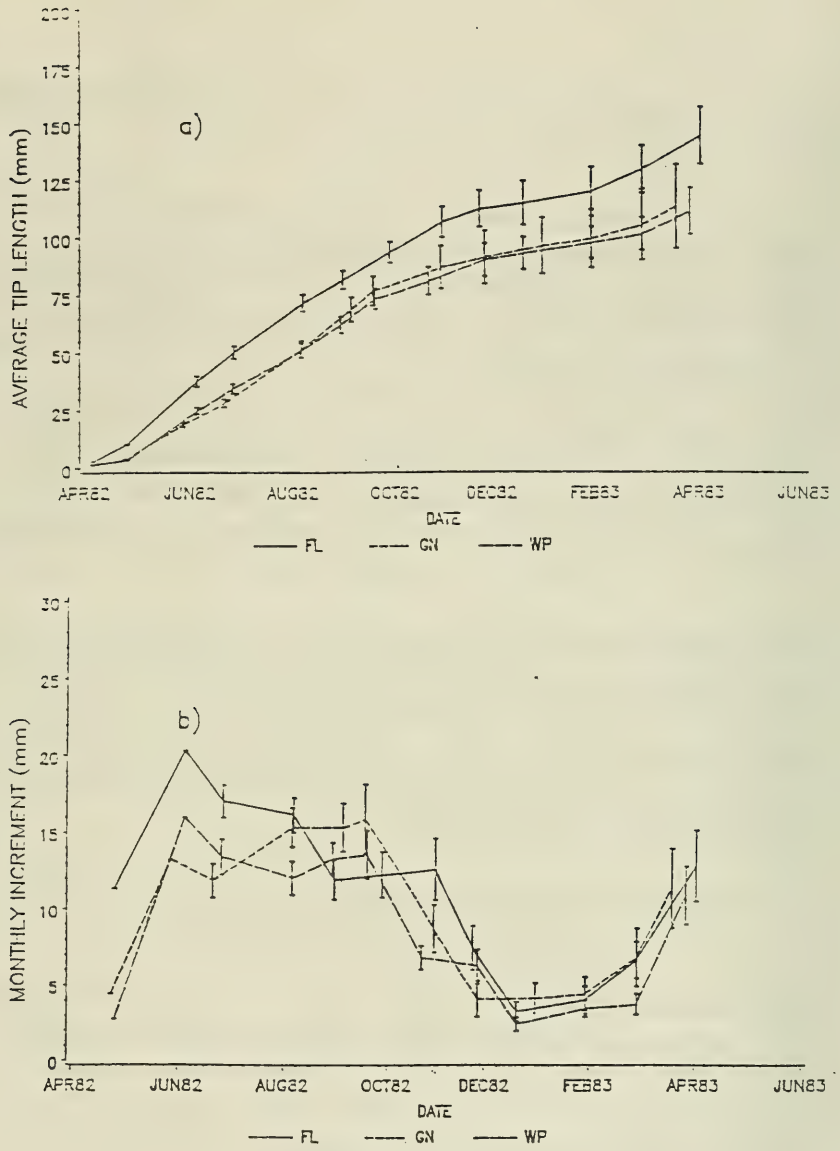

Figure 18. *Ascophyllum* growth, 1982-1983: a) as average tip length, and b) as average monthly increments. Data are plotted as means ±2 standard errors.

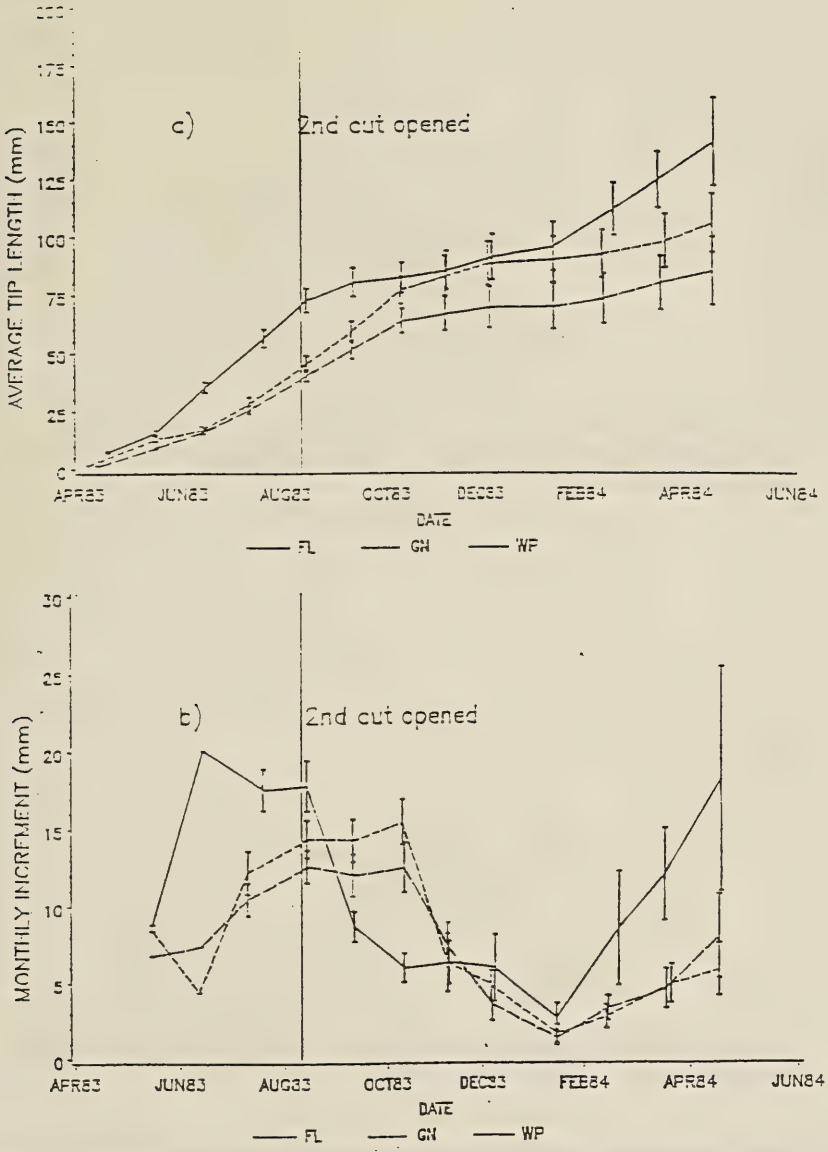

Figure 19. *Ascophyllum* growth, 1983-1984: a) as average tip length, and b) as average monthly increments. Data are plotted as means ±2 standard errors.

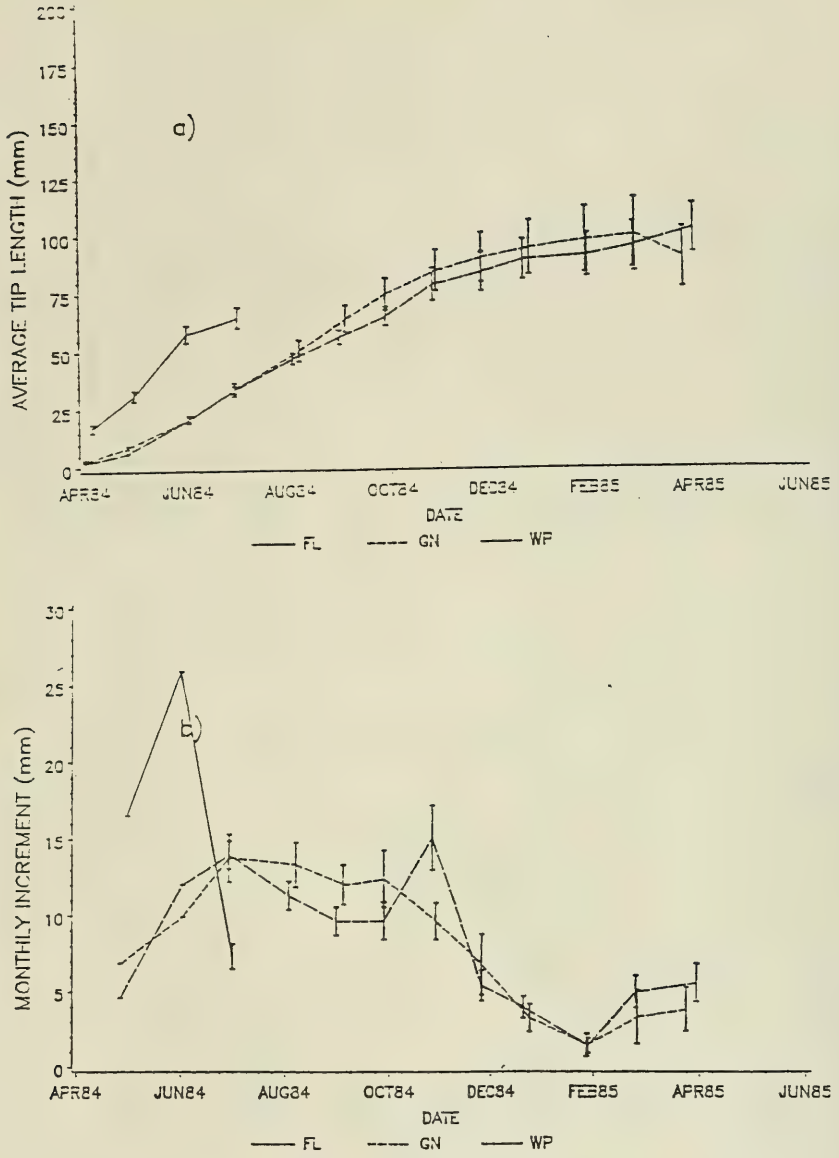

Figure 20. *Ascophyllum* growth, 1984-1985: a) as average tip length, and b) as average monthly increments. Data are plotted as means ±2 standard errors.

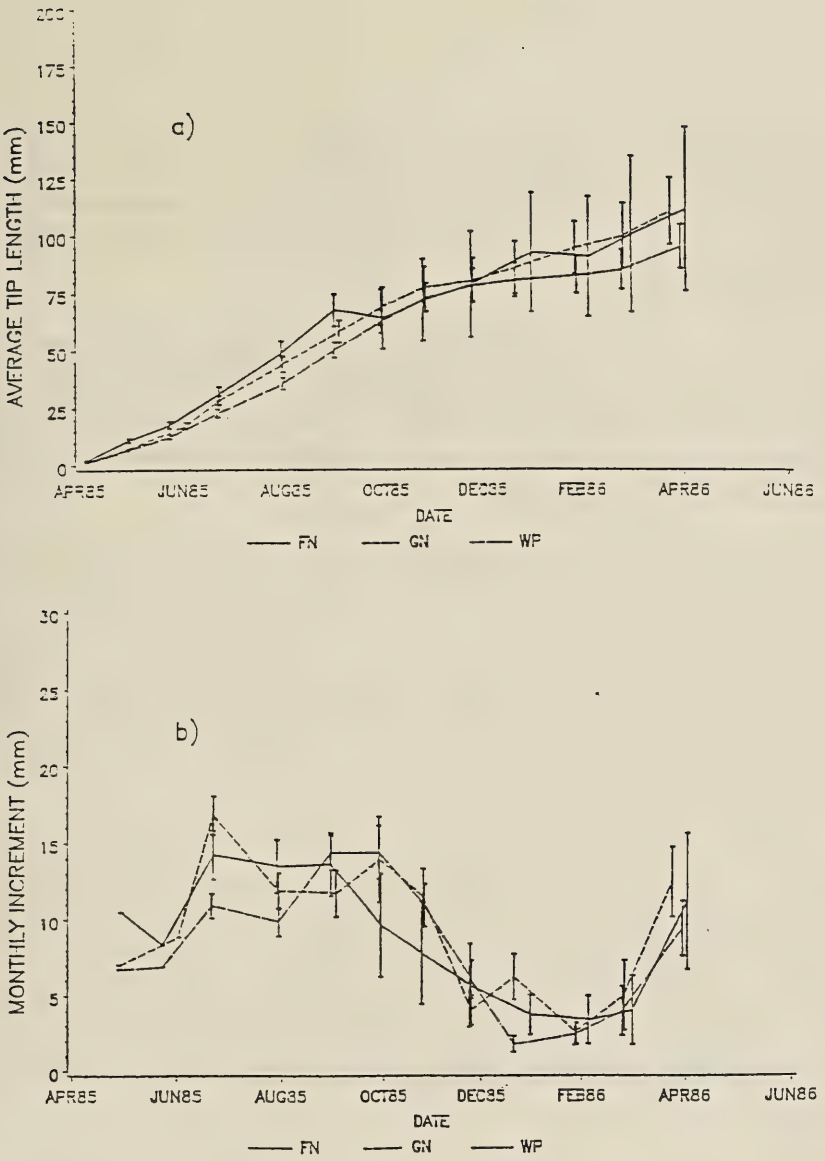

Figure 21. *Ascophyllum* growth, 1985-1986: a) as average tip length, and b) as average monthly increments. Data are plotted as means ±2 standard errors.

wrap used as a tip tag, or between the tip tag and the growing apex. To determine if the causes of mortality differed between stations, tip mortality was examined in two ways: as surviving tapes, and as surviving tapes with at least one viable apex (Figs. 22 and 23). Loss of tip tags implies mechanical removal and immediate loss of plant material. Loss of viable apices and/or damage to the apical cell implies a potential loss of biomass due to lack of growth.

Factors that contribute to *Ascophyllum* mortality include epiphytization (e.g., by *Polysiphonia lanosa*, *Ceramium rubrum*, *Ectocarpus siliculosus*) and grazing (especially by *Littorina obtusata*). Both of these processes increase the likelihood of breakage during storms. Thermal stress was evident only at FL < 1 month after the opening of the second quarry cut, and in the following year, tagged *Ascophyllum* plants were eliminated from FL by September 1984.

Since 1979, *Ascophyllum* plant loss averaged ca. 60% and tip loss ca. 80% (NUSCo 1986). In general, there was much year-to-year variability in plant and tip mortality. Prior to the opening of the second quarry cut, mortality was never associated with proximity to the MNPS discharge. Maintenance of stable populations, despite high measures for loss of plant material, is an indication of both the high degree of productivity of *Ascophyllum*, and the importance of this plant in contributing to the detrital pool of the marine ecosystem. Similar conclusions, and similar rates of mortality have been reported by other researchers in New England (Vadas et al. 1976; Wilce et al. 1978).

Ascophyllum is not expected to recolonize at FL, even though conditions may be favorable for most of each year. Since repopulation involves long-term survival of individuals (as discussed earlier for *Chondrus*), even short-term exposure to lethal water temperatures in summer prevents *Ascophyllum* recovery. The substratum previously occupied by *Ascophyllum* in this area will continue to be dominated by ephemeral algae. We must emphasize, however, the localized scale of this impact, i.e., less than 150 m of shoreline. Sampling of populations more distant to the discharge will continue to provide information to the rocky intertidal monitoring program. In particular, analyses of growth during 3-unit operation permits determination of whether thermal effects may be seen over a larger area.

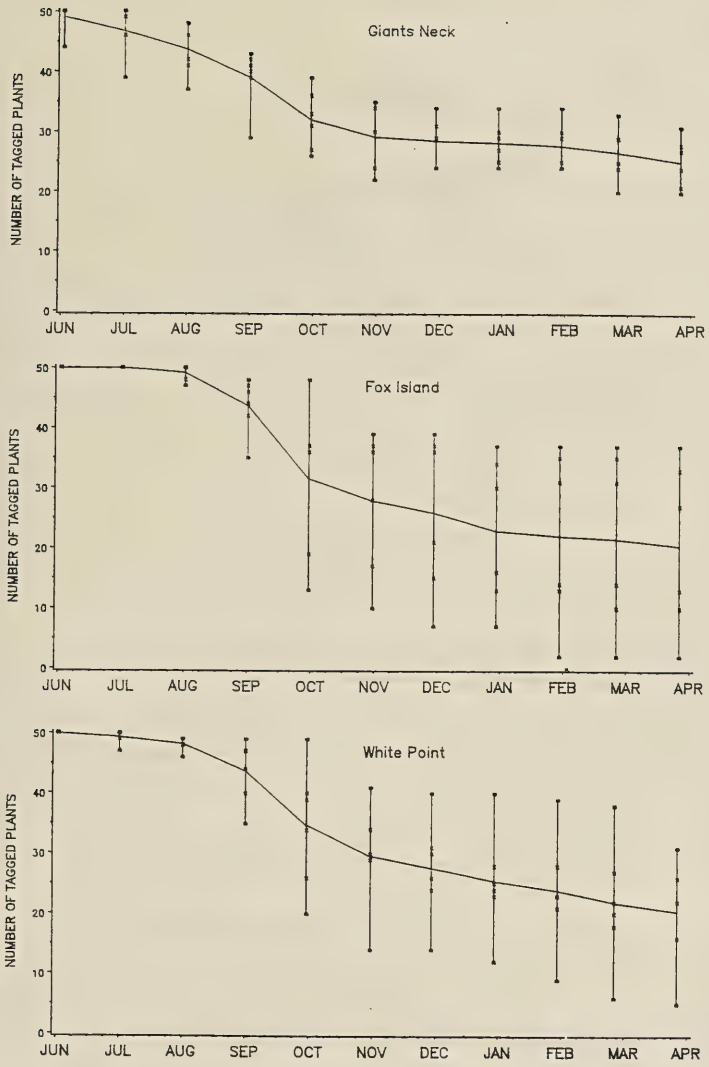

Figure 22. *Ascophyllum* mortality, as number of surviving tagged plants. Means of monthly values from 1979 to 1986 are plotted with their ranges.

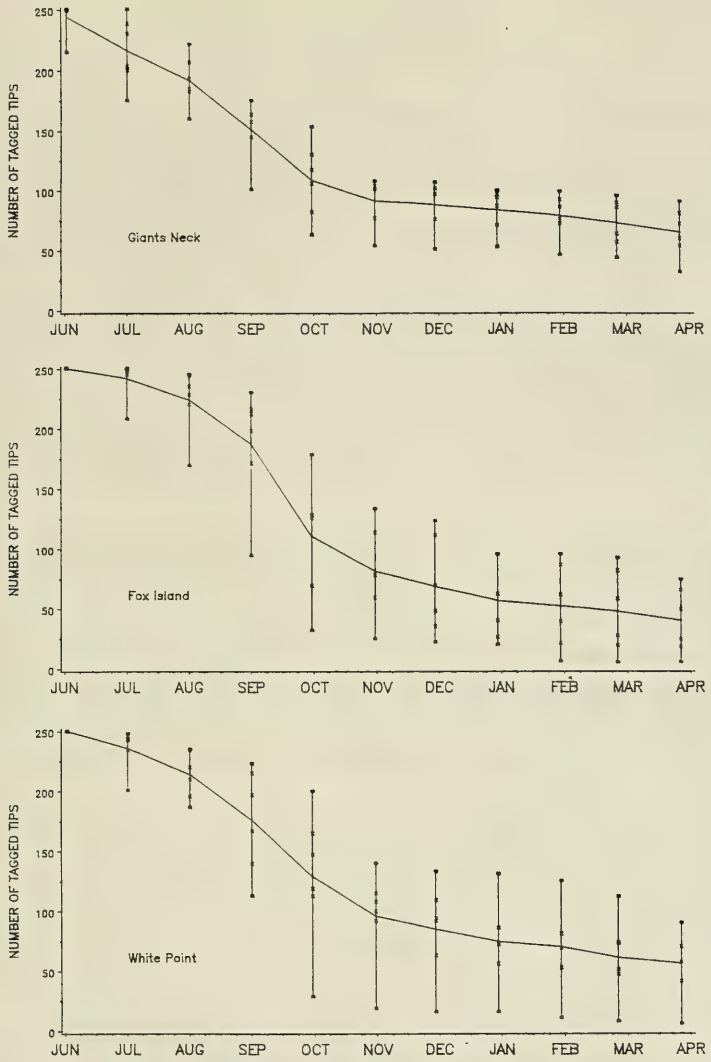

Figure 23. *Ascophyllum* mortality, as number of surviving tagged tips. Means of monthly values from 1979 to 1986 are plotted with their ranges.

SUMMARY

1. Water temperature is an important environmental parameter affecting local rocky intertidal communities. At sampling stations exposed to ambient water temperatures, communities closely resemble those studied throughout New England. The Millstone area flora has exhibited consistent patterns of spatial and temporal distribution during 2-unit operation. Since 1979, 158 species have been reported: 73 reds, 40 browns, and 45 greens. Quantitative studies show intertidal zonation patterns typical of rocky shores throughout New England: the high intertidal was dominated by barnacles, the mid intertidal by barnacles and fucoids, and the low intertidal was dominated by *Chondrus*. Degree of exposure to waves and storms played a determining role in the structure of rocky intertidal communities, either directly by minimizing desiccation and providing nutrients, or indirectly by influencing the distribution of grazers and predators. Changes to local rocky shore communities have been minor (except for the alteration at FE after the second cut opening) and predictable, based on naturally occurring cycles, indicating that environmental conditions during 2-unit operation have been stable.

2. The intertidal community at Fox Island-Exposed was unimpacted by 2-unit operation, prior to the opening of the second quarry cut in August 1983. Subsequent qualitative and quantitative collections show exceptions to the typical spatial and temporal trends noted elsewhere (and evident at FE previously). Physiological limits of many species were exceeded at 28 °C in late summer 1984, resulting in a decrease in species number at FE. Perennials were replaced by opportunistic species (ephemerals); a decrease in browns with a concomitant increase in greens with extended temporal patterns was reported. The resulting community resembled that found in the discharge quarry.

3. Recolonization experiments were undertaken to isolate factors that influence the structure of local rocky intertidal communities. In the Millstone area, recolonization was influenced by time of year in which denuding occurred, and it was related to degree of exposure and intertidal height. For example, recolonization was rapid in the high intertidal of an exposed station and slow in the low intertidal of a sheltered station.

4. Results from exclusion cage studies support conclusions made earlier, i.e., Fox Island-Exposed was unimpacted prior to the opening of the second quarry cut. Rates and patterns of recolonization, both with and without the influence of grazers and predators, were unaffected by proximity to the discharge.

5. *Ascophyllum nodosum* is an important biomonitoring tool in the rocky intertidal program because of its response to water temperature change and its mode of linear growth. Analyses of tip length have allowed us to distinguish between an impacted population exposed to water temperatures 2-3 °C above ambient and populations at two reference stations. Further, the response of the experimental *Ascophyllum* population, especially after the opening of the second cut, allows us to assess the effects of sublethal and lethal water temperatures.

CONCLUSION

Rocky intertidal studies performed during 2-unit operation show that local rocky shore communities are in a state of dynamic equilibrium; settlement, recruitment, and growth are balanced by senescence and removal (by both physical and biological interactions). Most of these processes are cyclic, and we have identified their temporal and spatial variability. We have also determined the effect of elevated water temperatures on these processes, and documented floristic and vegetational changes in an established community close to the MNPS discharge. If similar changes occur at stations more distant to the MNPS discharge, the NUEL rocky intertidal studies constitute a base-line from which to assess potential impacts associated with Unit 3 operation.

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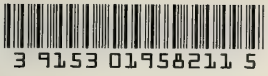
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Contents

BENTHIC INFAUNA 1
INTRODUCTION 1
MATERIALS AND METHODS 3
Chronology of Previous Sampling Protocols 3
Current Sampling Practices 5
DATA ANALYSES 7
Statistical Modeling 7
Biological Index Value 10
Species Diversity 11
Numerical Classification and Cluster Analyses 11
INTERTIDAL RESULTS 12
Sedimentary Environment 12
General Community Composition 12
Community Dominance 15
Community Abundance 17
Numbers of Species 20
Species Diversity 20
Cluster Analysis 23
INTERTIDAL DISCUSSION 25
INTERTIDAL CONCLUSIONS 26
SUBTIDAL RESULTS 27
Sedimentary Environment 27
General Community Composition 29
Community Abundance 31
Numbers of Species 31
Dominance 36
Species Diversity 39
Cluster Analysis 40
SUBTIDAL DISCUSSION 41
SUBTIDAL CONCLUSIONS 43
REFERENCES CITED 44
APPENDICES 51

BENTHIC INFAUNA

INTRODUCTION

The benthic infauna are relatively small inconspicuous organisms that inhabit intertidal beach and subtidal bottom sediments. Infaunal communities are generally composed of worms (polychaetes and oligochaetes), clams and small crustaceans and collectively, they play a vital role in maintaining ecosystem productivity. For instance, many infaunal organisms are important prey species for demersal fishes (Woodin 1982; Moeller et al. 1985; Witman 1985; Le Mao 1986) and thus form an important link in energy transfer pathways in the food chain. Their contribution to primary production in marine environments, though less conspicuous, is of equal importance. Many studies have described the influence of infaunal feeding, burrowing and tube building activities on nutrient recycling (Goldhaber et al. 1977; Aller 1978; Hylleberg and Maurer 1980; Raine and Patching 1980). The importance of this recycling process was documented by Zeitzschel (1980), who estimated that 30-100% of the nutrients required by shallow-water phytoplankton are derived from the sediment; the activities of the benthos often enhance this nutrient release.

Infaunal organisms are also useful environmental monitoring tools. These species are relatively sedentary and thus are often exposed to and cannot escape the anthropogenic stress. Further, the manner in which infaunal communities respond to stress is highly predictable (Boesch 1973; Reish 1973; Sanders et al. 1980; Boesch and Rosenberg 1982; Young and Young 1982; Rees 1984). For example, a physically stressed community is usually comprised of a few characteristically abundant species (e.g, *Polydora ligni*, *Capitella* spp. and *Mediomastus ambiseta*) which rapidly invade after disturbance or are capable of tolerating environmental stress (McCall 1977; Reish et al. 1980; Sanders et al. 1980).

To evaluate impacts based on the abundance and species composition of infaunal communities assumes that the organization of natural, undisturbed communities can be established, and that the direction and extent of any natural trends can be identified (Nichols 1985). Many studies have shown that cold winters (Beukema 1979), storms (Boesch et al. 1976), or heavy rainfall (Jordan and Sutton 1985; Flint 1985) and changes in such biological factors as competition and predation (Levinton and

Stewart 1982; Woodin 1982; Moeller et al. 1985) can strongly influence the abundance and composition of infaunal communities. As a result, the structure of natural infaunal communities oscillates around an equilibrium point and the range can only be established through long-term studies (Holland 1985).

The sampling of infaunal communities to identify environmental impacts associated with construction and operation of the Millstone Nuclear Power Station (MNPS) began in 1969, one year before start-up of Millstone Unit 1, has continued interrupted since that time. Environmental changes associated with construction and operation of MNPS include: bottom scour near the intake and discharge structures caused by water currents; chemical and heavy metal additions to the water; dredging near the intake and discharge structures and increased water temperatures due to cooling water discharge. To assess the degree of impact of these power plant induced changes on infaunal communities, the MNPS benthic monitoring program was designed to:

(1). Describe infaunal community abundance and composition at subtidal and intertidal stations located within and beyond areas influenced by operation and construction of the MNPS,

(2). Identify spatial and temporal patterns in community structure and establish the extent and direction of natural changes in these communities.

(3). Evaluate whether any observed changes in infaunal communities were the result of construction and/or operation of MNPS and; if so, assess their ecological significance.

Since 1975, the long-term infaunal program has centered on assessment of impacts that might occur during 2-unit operation. In recent years, additional short-term studies were implemented to provide data that will be used to assess nearfield impacts that result from Unit 3 start-up. These studies included an investigation of the infaunal community inhabiting the Millstone Discharge Quarry (Appendix BI-I) and a study of infaunal organisms inhabiting the area of the discharge cut (Appendix BI-II). The following report is intended to summarize results of the long-term sampling prior to the commercial operation of Unit 3 (April 1986) and is intended to establish a baseline for assessing impacts that might occur during 3-unit operation at the Millstone facility.

MATERIALS AND METHODS

Since the beginning of infaunal monitoring, numerous modifications in sampling procedures and laboratory techniques have been implemented to improve the quality of the data (Table 1). A chronological description of program changes are provided below; however, details of these changes, including justification for, and the overall effect on the data, are provided as appendices to this report.

Chronology of Previous Sampling Protocols

Sampling of infaunal organisms began in 1969, with studies of intertidal habitats. These studies provided comparisons between population sizes of the gem clam, *Gemma gemma*, at a potentially impacted station (Jordan Cove) and a reference station (Giants Neck). This program also provided meiofaunal biomass estimates; however, organisms were not identified. Sampling methods used during these studies (May 1969 through March 1973) were developed to obtain abundance estimates of primarily one species, thus community parameters that are currently used to assess environmental impacts can not be calculated. These data have been summarized elsewhere (Hillman et al. 1973) and will not be considered further in this report.

In March 1973, the scope of the benthic infaunal program was expanded and reflected adoption of a "community" approach to assess environmental impacts rather than an "indicator species" approach. In addition, sampling of subtidal habitats began and an additional intertidal station was established.

Since 1973, the "community" approach to evaluating power plant impacts has continued, although modifications in sample size, numbers of replicates, collection locations and schedules, preservation methods, and mesh size used to process samples have influenced the data. For instance, the practice of freezing samples, (from 1973 to June 1976), resulted in the loss of up to 75% of the organisms (NUSCo 1982). These data can not be quantitatively compared to subsequent data and will not be used in any future study of potential impacts of the Millstone facility.

Infaunal data collected in June and September 1976 were analyzed to assess the adequacy of ten replicate cores in sampling infaunal communities at each station (Battelle 1977). Results identified the patchy distributions of infaunal organisms and concluded with a recommendation that increased replication

Table 1. Chronological summary of Field and laboratory protocols used to collect subtidal and intertidal infaunal samples during monitoring studies at Millstone Nuclear Power Station from May 1960 - March 1986.

INTERTIDAL

May 1969	May 1973	Jun 1973	June 1974	Sep 1974	Mar 1976	Sep 1976	Dec 1978	Mar 1979	Mar 1983 - Mar 1986
/5 cores 12.75-cm I.D. x5-cm deep		5 Boxes 25x25x5 cm		10 cores 10-cm I.D. x5-cm deep				15 cores	10 cores
/70% ethanol (EtOH)				frozen, then 70% EtOH		-70% EtOH		10% Formalin, then EtOH	
/2.0, 1.0, 0.4-mm mesh sieve				0.71-mm. mesh sieve				0.5-mm mesh sieve	
/Feb, May, Jul, Sep, Dec			Mar, Jun, Sep, Dec						
/Jordan Cove									
/Giants Neck-mouth of small stream, east of seining beach			west end of seining beach						
			/White Point						

SUBTIDAL

Mar 1973	Mar 1974	Jun 1974	Mar 1975	Jun 1976	Sep 1976	Apr 1977	Jun 1978	Dec 1978	Mar 1979	Mar 1983-Mar 1986
/5 boxes 25x25x5 cm		10 cores 10 cm I.D. x 5-cm deep							15 cores	10 CORES
/Frozen then 70% ethanol (EtOH)				70% EtOH				10% Formalin, then EtOH		
/0.71-mm mesh sieve									0.5-mm mesh sieve	
/Mar, Jun, Sep, Dec										
/Jordan Cove, Effluent										
/Two Tree Channel, Little Rock, Bay Point										
	/ Intake									
		/ Giants Neck								
				/ Niantic Bay						
						/ Seaside				

might help integrate small scale patchiness and thus provide better estimates of density, species diversity and patterns in community composition. Following this report, a more intensive study examined the effects of sample size on the various parameters used to characterize infaunal communities (e.g., species area curves, indices of dispersion, species diversity, abundance, dominance patterns) (Battelle 1978). Results showed that larger cores (25-cm (i.d.) x 10-cm deep) were more effective than smaller cores (10-cm (i.d.) x 5-cm deep) in collecting less abundant species, although significant differences in density were evident for only three species. Based on this study, a recommendation for increasing the numbers of replicate cores from 10 to 15 was implemented in March 1979. In addition, samples were processed with a 0.7 mm and a 0.5 mm mesh sieve to provide a comparison of the numbers of individuals retained by each sieve. Because of the increased numbers of replicates, several sampling stations were eliminated and use the 10-cm core was adopted to allow sufficient time to process the additional samples.

In 1981, data collected during the previous two years were analyzed to evaluate the effects of 15 vs 10 replicates (NUSCo 1982). This study revealed that 10 cores were needed to collect 90% of the species found in 15 cores. Species composition was similar using both methods, overall estimated densities were not significantly different, and the estimates of community variance were not substantially lower based on 15 replicates. Based on this study, the replicate number was reduced to 10 cores per station/quarter. As might be anticipated, use of the smaller mesh sieve significantly increased both the numbers of individuals and species collected (NUSCo 1986).

Current Sampling Practices

From March 1979 to March 1986, infaunal communities were sampled at four subtidal and three intertidal stations (Fig. 1). The Giants Neck subtidal (GN-S) and intertidal (GN-I) stations are located 5.5 km west of the power plant and serve as reference stations. The Intake subtidal station (IN-S) is located 0.1 km seaward of the Millstone Unit 2 intake structure and the Effluent subtidal station (EF-S) is approximately 0.1 km offshore and adjacent to the cooling water discharge into Long Island Sound. The EF-S station is located as close to the effluent as possible given the current produced by the discharge. Jordan Cove subtidal (JC-S) and intertidal (JC-I) stations are located 0.5 km east of the power plant and based on thermal plume mapping studies (and modeling of Unit 3) (NUSCo 1983) are located in areas potentially influenced by the plant discharge during two and three-unit operation. The

Figure 1. Map of the Millstone Point area showing the location of intertidal and subtidal sand sampling stations. (GNI, JCI and WPI = Giants Neck, Jordan Cove and White Point intertidal stations; GNS, EFS, JCS, INS = Giants Neck, Effluent, Jordan Cove, and Intake subtidal stations).

White Point (WP-I) intertidal station is farther east of the power plant (1.6 km), but is still within an area potentially influenced by plant discharge, particularly when all three units are operating.

At each subtidal and intertidal station, ten 0.0078 m^2 cores (10 cm diameter x 5 cm deep) were collected quarterly (March, June, September and December). For reporting purposes, a sampling year begins in September and ends in June, and the year of the June sample assigned as the sampling year. Subtidal samples were taken within 3 m of each station marker by SCUBA divers. Each sample was

placed in a 0.333 mm mesh Nitex bag and brought to the surface. Intertidal samples were collected at approximately 0.5 m intervals parallel to the water line at mean low water.

Samples were brought to the laboratory and fixed with a 10% buffered formalin/rose bengal solution. After a minimum of 48 h, organisms were floated from the sediments onto a 0.5 mm mesh sieve and the float and residue were preserved separately in 70% ethyl alcohol. Organisms were removed under dissecting microscopes, sorted into major groups (annelids, arthropods, molluscs, and others), identified to the lowest possible taxon and counted. Taxa not quantatatively sampled by our methods, (e.g., nematodes, ostracods, copepods, and foraminifera) were not removed from samples.

A 3.5-cm diameter x 5-cm deep core was taken at the time of infaunal sampling and sediment analysis performed using the dry sieving method (Folk 1974). His method of moments technique was used to calculate the arithmetic mean phi, which was then converted to mean grain size.

DATA ANALYSES

This report includes data that were collected from March 1979 through March 1986, the last sampling period prior to commercial operation of Millstone Unit 3. Quarterly results from this period are presented either on Tables within the text or attached as Appendices. When analyses are based on annual means or annual totals, only data comprising an entire sampling year (e.g., the September, December, March and June quarters) are used. In this report, data collected in March and June 1979, September and December 1985 and March 1986 were not used in analyses involving annual means because they do not constitute an entire sampling year.

Statistical Modeling

Multiple regression techniques were used to minimize natural sources of variation and thus improve the sensitivity of analyses designed to identify year to year differences in animal abundance and numbers of species. Data used were log-transformed quarterly mean densities (no./core) and quarterly mean

numbers of species (no./core) collected from March 1979 through March 1986. As noted previously, natural temporal and spatial fluctuations in infaunal communities often reflect environmental changes due to climatic conditions or to life history cycles of the organisms comprising these communities. To remove this natural variation, the following were used as explanatory variables in our multiple regression models:

A. Precipitation

Daily precipitation records compiled by the U.S Weather Bureau at the Groton Filtration Plant were obtained from June 1976 through March 1986. Values to the nearest 0.01 inch were used as our "rain" data for the regression model.

B. Water and Air Temperature

Ambient water temperatures (at the Millstone Intake Structures) and air temperatures (at the 33-foot level of Millstone meterological tower) were extracted from the Northeast Utilities Environmental Data Acquisition Network (EDAN). Daily averages, based on observations made at 15-minute intervals, were calculated for the period June 1976 to March 1986.

C. Wind Speed and Direction

Wind speed and direction (at the 33-foot level of the Millstone meterological tower) were extracted from the EDAN database at 15-minute intervals from June 1976 to March 1986. These values were used to calculate a Wind Index which weighted wind speed according to wind direction. A NOAA navigational chart of the sampling area was used to calculate site-specific wind direction weights according to the particular wind direction responsible for potential wave-induced sediment disturbances at each sampling station. The directional weight ranged from 0, when wind would not influence the station, to 1, when the wind could result in waves directly affecting the area. The Wind Index was then computed by multiplying the directional weight times the wind speed. Because the effect of wind was assumed to be cumulative, daily averages were derived using only Wind Index values greater than 0 (i.e., wind direction and speed during the 15-minute periods when such wind-induced effects could have occurred).

D. Sedimentary Parameters

Sedimentary parameters (e.g., mean grain size and silt/clay content) were obtained as part of the monitoring studies and the quarterly values for each used as explanatory variables in the regression analyses.

E. Climatic Extremes (Deviations)

Additional variables were created to represent periods of extreme climatic conditions which have occurred during the sampling period. High or low deviations were derived for each abiotic factor as the difference between the quarterly mean or daily value and the ten year mean for that quarter. Deviations based on quarterly means were intended to examine the effects of longer term extremes (i.e., an unusually cold winter), while those based on daily values were intended to remove the effects of shorter-term episodic events (i.e., storms). Daily deviations were averaged and summed (for cumulative effects) over each sampling quarter.

F. Seasonal Reproduction-Recruitment Component

Many infaunal organisms in the Millstone area exhibit annual peaks in abundance, which often reflect the seasonal nature of annual reproduction and recruitment cycles or periods of favorable climatic. Spectral analyses of quarterly data showed that annual cycles in community abundance and numbers of species were present. To account for this periodicity, harmonic terms having a period of 1 year were included as explanatory variables in the regression models.

In all, 32 abiotic variables were available for the analyses (28 climatic variables, 2 sedimentary charateristics and 2 seasonal harmonic components).

Model Selection Procedure

Long-term trends in the quarterly mean values were first detrended using a polynomial regression equation. If no significant long-term trend was evident, residuals were created by subtracting the quarterly mean from the six-year mean. A stepwise multiple regression on these residuals was then used to select

relevant abiotic variables and combinations of variables that were significant at a probability level of $\alpha < 0.15$. This probability was deemed sufficient to guard against fitting more parameters than can be reliably estimated, given the sample size. The model that minimized the mean square error and maximized the R-square was selected as best describing observed variablity in abundance and numbers of species.

Analyses of covariance were then conducted to test for annual differences in abundance and species using significant explanatory variables as covariates. Results of these analyses and pair-wise t-tests on adjusted means (least square means) were used to identify significant ($\alpha < 0.05$) long-term and annual differences species abundance and number.

Although some abiotic factors were not independent of each other, all variables were initially included in the stepwise regression analyses, because our objective was to identify and remove the natural temporal variability and thus improve our ability to detect power plant related changes. Future analyses of data collected during 3-unit operation will be directed to identifying the extent and effect of individual abiotic factors in structuring local infaunal communities.

Biological Index Value

The Biological Index Value (BIV) of McCloskey (1970), an index of dominance, calculated using annual totals of the 10 most abundant taxa at each station collected from 1980-1985. Species were ranked according to their total abundance in each sampling year and the ranks summed for all years. To calculate the BIV, the sum for each taxon was expressed as a percentage of a theoretical maximum sum that would occur if a species ranked first in all sampling years. For example, the BIV would be equal to 100% and the theoretical maximum equal to 60 when a species ranks first in abundance in each of six years and a total of 10 species are collected.

Species Diversity

Species diversity for each station was calculated using the Shannon information index:

$$H' = \sum_{i=1}^{S} \frac{n_i}{n} \frac{\log_2}{N} \quad (\textit{Pielou } 1977)$$

where n_i = number of individuals of the i^{th} species, N = total number of individuals for all species and S = number of species. An evenness component of diversity was calculated as:

$$J = \frac{H'}{H_{max}} \quad (\textit{Pielou } 1977)$$

where H_{max} = log S and represents the theoretical maximum diversity when all species are equally abundant. Evenness ranges from zero to one and increases as the numbers of individuals among species become more evenly distributed. Diversity calculations excluded oligochaetes and rhynchocoels (groups that sometimes accounted for over 80% of the totals organisms collected) because they were not identified to species. Similarly, other organisms that could not be identified to species, either because they were juveniles or in poor physical condition, were excluded from this analysis.

Numerical Classification and Cluster Analyses

Cluster analyses, based on annual abundances of organisms were performed using the Bray-Curtis similarity coefficient. This coefficient is calculated as:

$$S_{jk} = \frac{\sum_i 2\min(X_{ij}, X_{ik})}{\sum_i (X_{ij} + X_{ik})} \quad (\textit{Clifford and Stevenson } 1975)$$

where X_{ij} = abundance of attribute i at entity j and X_{ik} = abundance of attribute i at entity k. Based on these similarities, cluster analyses incorporating a flexible sorting strategy (B = -0.25) was used to form station groups (Lance and Williams 1967).

INTERTIDAL RESULTS

Sedimentary Environment

The JC station is a semi-protected, southeasterly facing beach that is seasonally exposed to waves produced by southeast winds (primarily during fall and winter storms). The sediment at this station is composed of medium to very coarse sands; mean grain size since March 1979 ranged from 0.3 - 1.3 mm (Fig. 2). These sediments generally contain only 1-3% silt/clay and large amounts of eelgrass (*Zostera marina*) and algae often cover the beach. The beaches at GN and WP face southerly and are exposed to the prevalent south to southwesterly winds which occur in the Millstone area. Wave scour produces clean, sandy beaches composed of medium sand which ranged from 0.3 to 0.8 mm in size since 1980; silt/clay content at these stations has been consistently low (< 1%).

During the monitoring period, temporal fluctuations in sediment grain size and the percentage of silt/clay have been characteristic of the JC station and possibly reflect the more seasonal nature of erosion and accretion cycles. In contrast to JC, sedimentary characteristics of the GN and WP beaches have exhibited temporal stability, both seasonally and annually and is probably due to more uniform exposure to wave induced scour.

General Community Composition

The 720 samples collected at the three intertidal beaches during the baseline period yielded 135 taxa and 69,448 individuals. Since 1980, communities have been dominated by polychaetes and oligochaetes, which frequently accounted for over 75% of the total number of individuals collected annually (Table 2). The JC community was dominated by the oligochaetes, which accounted for 40-86% of the individuals collected in each of the last six years. At GN and WP, polychaetes were generally most abundant and oligochaetes accounted for less than 40% of the individuals. At these stations, rhynchocoels were often an abundant component of the community. In terms of species number, polychaetes were most numerous and, although arthropod and mollusc species were present at JC and GN, these groups accounted for less than 5% of the total individuals collected.

Figure 2. Quarterly mean grain size (mm) and silt/clay content (%) of sediments sampled at Millstone intertidal stations from March 1979 - March 1986.

Table 2. Number of species (S), number of individuals (N) and relative percent of total (%) for each major taxon collected at Millstone intertidal stations sampled from September 1979 - June 1985.

	1980			1981			1982			1983			1984			1985		
	S	N	%	S	N	%	S	N	%	S	N	%	S	N	%	S	N	%
Giants Neck																		
Polychaeta	24	1819	61	18	1649	73	13	2222	82	15	1036	67	24	908	60	14	613	40
Oligochaeta	-	191	6	-	79	4	-	172	6	-	143	9	-	458	30	-	626	40
Mollusca	8	14	1	3	3	<1	1	10	<1	0	0	0	5	13	1	1	1	<1
Arthropoda	11	22	1	5	26	1	14	59	2	5	12	1	8	20	1	6	12	<1
Rhynchocoela	-	932	31	-	498	22	-	260	10	-	362	23	-	109	7	-	297	19
Totals	43	2978		26	2255		28	2727		20	1553		37	1508		21	1549	
Jordan Cove																		
Polychaeta	22	1686	12	20	4120	30	21	1055	16	19	2631	17	12	2176	56	20	2133	12
Oligochaeta	-	12156	85	-	8666	63	-	4749	70	-	12430	81	-	1582	40	-	15012	86
Mollusca	9	112	1	9	711	5	6	10	<1	7	26	<1	10	93	2	8	67	<1
Arthropoda	12	265	2	16	100	1	12	968	14	17	254	2	12	55	1	24	175	<1
Rhynchocoela	-	64	<1	-	70	1	-	23	<1	-	1	<1	-	8	<1	-	17	<1
Totals	43	14283		45	13667		39	6805		43	15342		34	3914		52	17404	
White Point																		
Polychaeta	21	852	56	17	1238	54	22	1548	60	12	428	48	15	746	44	16	714	48
Oligochaeta	-	59	4	-	273	12	-	669	26	-	107	12	-	277	16	-	460	31
Mollusca	4	5	<1	3	3	<1	1	1	<1	1	3	<1	4	4	<1	6	18	1
Arthropoda	0	2	<1	2	5	<1	3	6	<1	4	5	<1	4	11	1	7	10	1
Rhynchocoela	-	594	39	-	783	34	-	337	13	-	356	40	-	679	40	-	273	19
Totals	25	1512		12	2302		26	2561		17	895		23	1717		29	1475	

Quarterly numbers of species, total numbers of individuals and relative abundances from September 1979 to March 1986, for each major taxon collected at intertidal stations are presented in Appendix I. Highest abundances of polychaetes and oligochaetes generally occurred in September or June. Molluscs and arthropods, more common in the JC community, have also been most abundant in either September or June. Rhynchocoels, an important component of the GN and WP communities, were generally most abundant in colder months (December and March).

In the three 1986, sampling periods (September/December 1985 and March 1986), there were temporal shifts in general community composition evident at all intertidal stations. For instance, polychaete species number at WP and GN stations was lower than all previous observations in the September collections. The total number of individuals at WP in September was lower than most previous observations. The GN and WP stations also had relatively low total numbers of individuals in December and March collections, lower numbers of polychaete species and total numbers of species in the December collections. Although the three JC sample periods were similar to most previous sampling periods, the total number of individuals in December was sightly lower. The JC 1986 March collection was among the highest for total numbers of species and individuals recorded for that period.

Community Dominance

Since 1980, intertidal communities have been numerically dominated by species of polychaetes, oligochaetes and rhynchocoels, and during this period, only one one arthropod (*Gammarus lawrencianus*) and one mollusc (*Gemma gemma*) species have accounted for more than 5% of the total individuals collected (Table 3). Only *Scolecolepides viridis*, *Polydora ligni*, and oligochaetes were among the dominants at all stations. At GN and WP, 9 of 10 numerical dominants were the same over the baseline period. In contrast, six of the top ten most abundant taxa at JC were dominants at only this station.

Sandy beach communities in the Millstone Point area were typically comprised of a few taxa that occurred in high densities over all sampling years, along with less abundant forms that exhibited temporal variations over the monitoring period. This was particularly true at JC, where oligochaetes accounted for over 60% of all individuals in 5 of the past 6 years and was the most abundant taxon in each of the last six sampling years (BIV = 100%). *Scolecolepides viridis* and *Hediste diversicolor* were the only

Table 3. Percent contribution and Biological Index Value (BIV) of the ten most numerically abundant taxa at Millstone intertidal stations in each year, September 1970 - June 1985.

Station	1980	1981	1982	1983	1984	1985	BIV
Giants Neck							
Rhynchocoela	32	22	10	23	7	19	90.2
Haploscoloplos fragils	15	26	10	11	12	19	89.2
Scolecolepides viridis	20	18	21	9	28	5	85.3
Paraonis fulgens	19	13	40	37	4	2	84.3
Oligochaeta	6	4	6	9	30	40	83.3
Capitella spp.	1	11	4	6	8	9	75.5
Hediste diversicolor	<1	5	2	<1	<1	2	53.9
Polydora ligni	2	<1	1	0	4	1	52.9
Microphthalmus sczelkowii	<1	1	<1	<1	<1	1	47.1
Tharyx acutus	1	<1	<1	1	0	0	36.8
Gammarus lawrencianus	<1	<1	1	<1	<1	<1	35.8
Streptosyllis arenae	0	0	1	0	1	<1	32.9
Neohaustorius biarticulatus	0	<1	<1	<1	<1	<1	32.8
Haustorius canadensis	1	<1	0	<1	0	<1	31.9
Pygospio elegans	<1	<1	1	0	<1	0	27.9
Lepidonotus squamatus	0	0	0	0	1	0	20.6
Polydora socialis	0	0	12	0	0	0	19.6
Jordan Cove							
Oligochaeta	87	64	70	81	40	86	100.0
Scolecolepides viridis	8	6	13	10	32	4	94.4
Hediste diversicolor	2	20	1	1	21	4	88.1
Polydora ligni	<1	1	<1	3	2	1	77.4
Capitella spp.	1	2	1	2	1	2	77.0
Gemma gemma	1	5	<1	<1	1	<1	68.3
Gammarus lawrencianus	<1	<1	14	1	<1	<1	65.1
Rhynchocoela	<1	<1	<1	<1	<1	<1	60.7
Gammarus mucronatus	<1	<1	<1	<1	<1	<1	53.2
Streblospio benedicti	<1	<1	<1	1	<1	<1	52.4
Microphthalmus sczelkowii	<1	<1	<1	<1	0	<1	44.0
Lacuna vincta	<1	0	<1	<1	<1	<1	41.3
Pygospio elegans	<1	0	0	5	<1	<1	40.9
Nereis succinea	0	<1	<1	0	0	<1	38.9
Leptocheirus pinguis	0	0	<1	1	<1	<1	35.3
Eteone longa	<1	0	0	0	0	0	34.9
Crepidula plana	<1	0	<1	0	<1	<1	31.3
Edotea triloba	0	<1	<1	<1	<1	<1	31.0
Phoxocephalus holbolli	0	0	<1	0	<1	0	22.2
Potamilla reniformis	0	0	0	0	0	<1	21.8
White Point							
Rhynchocoela	39	34	13	40	39	19	95.6
Haploscoloplos fragilis	21	21	15	31	23	11	92.1
Paraonis fulgens	8	12	34	7	15	26	89.5
Oligochaeta	4	12	26	12	16	31	88.2
Streptosyllis arenae	4	4	6	7	1	3	77.2
Capitella spp.	9	2	<1	<1	<1	4	66.7
Scolecolepides viridis	4	7	<1	<1	<1	1	60.5
Parapionosyllis longicirrata	<1	1	1	2	<1	<1	53.1
Exogone hebes	<1	<1	<1	<1	<1	<1	46.1
Polydora ligni	4	2	0	0	<1	1	45.6
Pygospio elegans	0	2	<1	<1	0	<1	42.1
Hediste diversicolor	1	<1	1	0	0	<1	38.6
Aricidea catherinae	0	0	2	0	<1	<1	37.3
Mytilus edulis	0	0	<1	<1	<1	<1	33.3

other taxa that had relatively high BIV's (> 80%) and consistently accounted for over 5% of individuals during the period. And although *Capitella* spp., *Polydora ligni*, and *Gemma gemma* have consistently ranked among the dominants, they usually were found in low densities relative to other the top three taxa.

At GN and WP, rhynchocoels, *Paraonis fulgens*, and *Haploscoloplos fragilis* were dominant, contributing over 10 percent of the total individuals and having BIV's over 80% indicating their consistent abundance over years. In addition to the above taxa, *Scolecolepides viridis* was a consistent dominant at GN and *Streptosyllis arenae* at WP.

Community Abundance

Quarterly mean abundance (no./core) of intertidal communities along with predicted values from multiple regression models are given in Figure 3. From March 1979 to March 1986 infaunal abundance (the exponential of values in Figure 3) ranged from 4 to 95 at GN; 18 to 840 at JC; and from 2 to 143 at WP. Overall, the JC community included more organisms, than GN and WP which were of similar magnitude. On a seasonal basis, intertidal organisms were generally more abundant during warmer sampling periods (June and September) than during colder ones (March or December). However, no consistent trend was common to all stations.

Multiple regression models removed 59%, 20% and 60% of the variability in abundance at GN, JC and WP, respectively. Plots of annual means after adjusting for this natural variation are presented in Figure 4. The analysis of variance, based on adjusted annual abundances revealed that no significant long-term trends in abundance have occurred at any intertidal station between 1980 and 1985.

Pairwise comparisons, based on t-tests, indicated that the annual mean abundance at JC during 1982 was significantly lower than those observed in 1983 and 1985. At GN, the 1984 value was significantly higher than those of 1981 or 1983. At WP, annual density in 1983 was significantly lower than those found 1981 or 1982; 1985 density was also significantly lower than 1982;

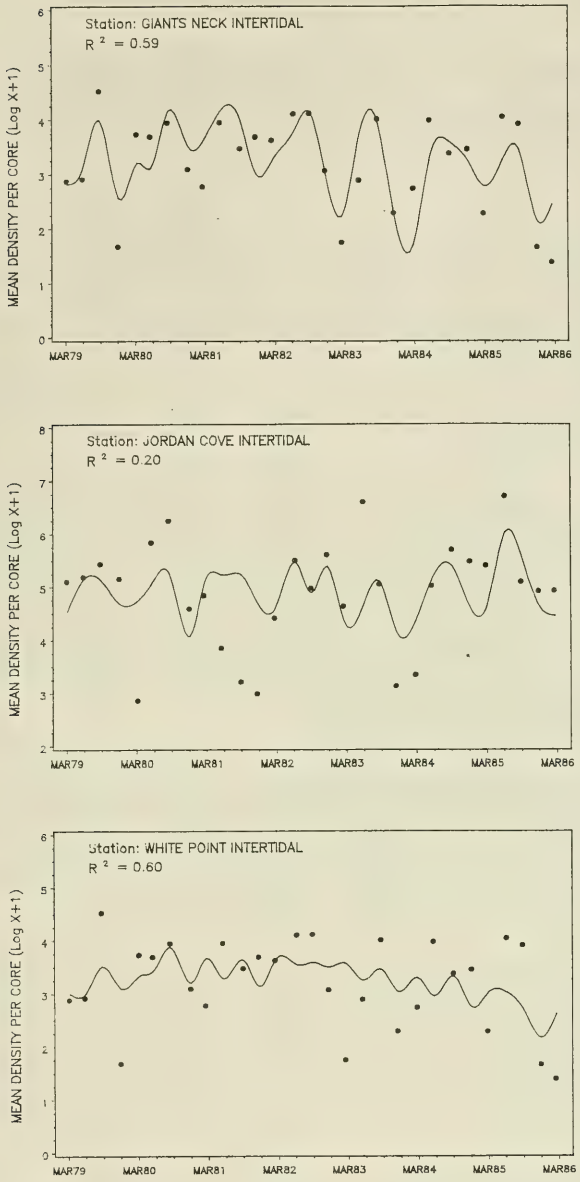

Figure 3. Quarterly mean numbers of individuals per core and multiple regression predictions for Millstone intertidal infaunal communities sampled from March 1979 - March 1986.

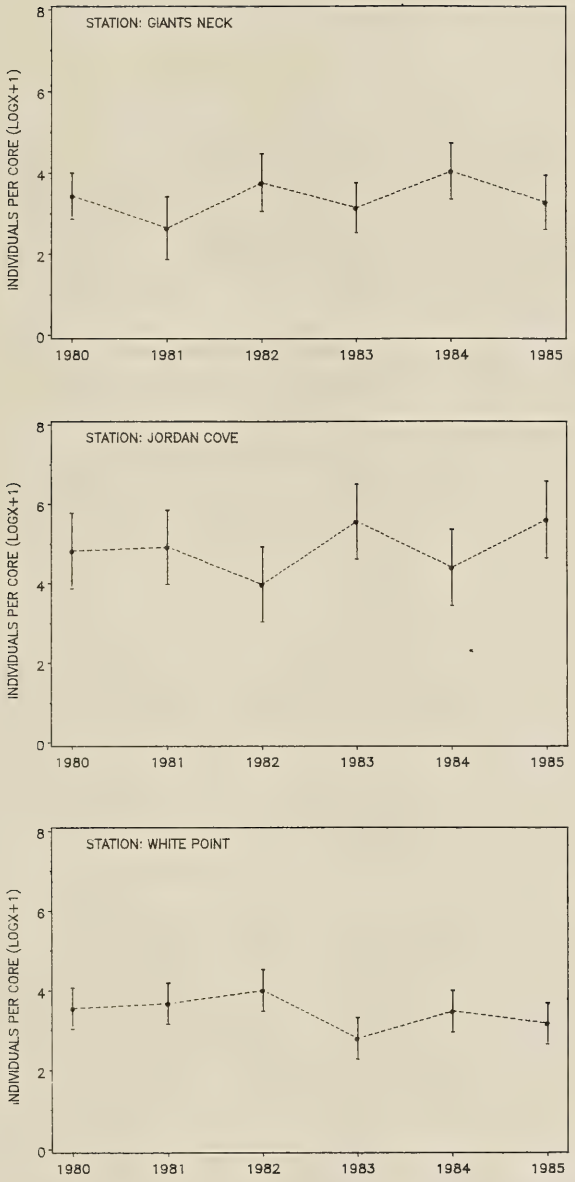

Figure 4. Annual mean abundance per core with two standard errors for intertidal infaunal communities sampled from September 1979 - June 1985. (Annual means were adjusted using analysis of covariance which included abiotic and climatic conditions as covariates.)

Numbers of Species

Quarterly mean numbers of species (no./core) from March 1979 - March 1986 ranged from 1 to 10 at GN; 3 to 13 at JC; and < 1 to 9 at WP (Fig. 5). As with overall community abundance, the number of species was highest at JC, while at GN and WP numbers of species were similar. The seasonal trend in the numbers of intertidal species was similar to that of density, being higher in September and June than in March or December.

Abiotic variables used in the multiple regression models accounted for 60% (GN), 63% (JC) and 62% (WP) of the total variation in species numbers. After removing this variation, no long-term trends in species numbers were evident at any station since 1980. Plots of annual adjusted mean numbers of species illustrated the stability of this community parameter during the monitoring period (Fig. 6). In addition, t-tests of adjusted annual means identified significant interannual differences between the 1982 and 1984 values at JC; all other pairwise comparisons between years were not significant.

Species Diversity

Annual mean species diversity (H′) over the study period ranged from 1.2 to 2.3, and evenness (J) from 0.4 to 0.7 (Table 4.). The mean numbers of species and individuals included in diversity calculations ranged from 6 to 17 and 97 to 580, respectively. On an annual basis, no consistent spatial or temporal shifts in H′ have occurred over the monitoring period. Other parameters (S, J, N), reflected spatial differences in the structure of intertidal communities. For example, at GN and WP both the numbers of species and individuals was generally lower than at JC.

Quarterly values of species diversity from September 1979 to March 1986 for intertidal stations are presented in Appendix II. Highest values of H′, S and J commonly occurred in September at all stations. For the three sample periods in 1986, H′, S and J attained historical low values in the GN September and December collections. At JC, H′ and J were low in December relative to previous values of these indices. Given the temporal variation of diversity parameters at WP, the values in the 1986 collections were considered similar to previous years.

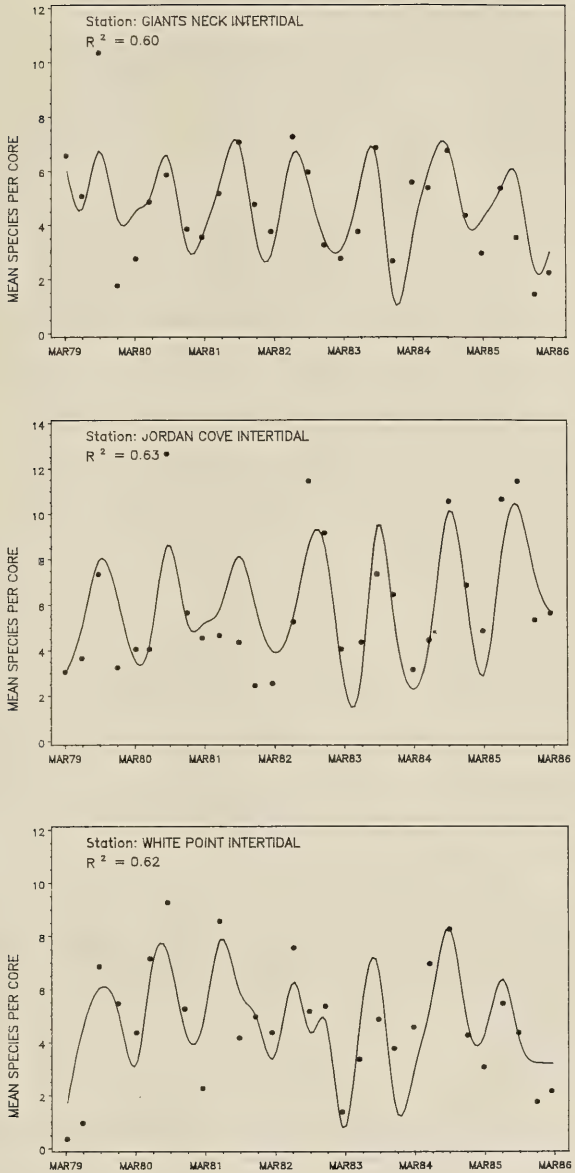

Figure 5. Quarterly mean numbers of species per core and multiple regression predictions for Millstone intertidal infaunal communities sampled from March 1979 - March 1986.

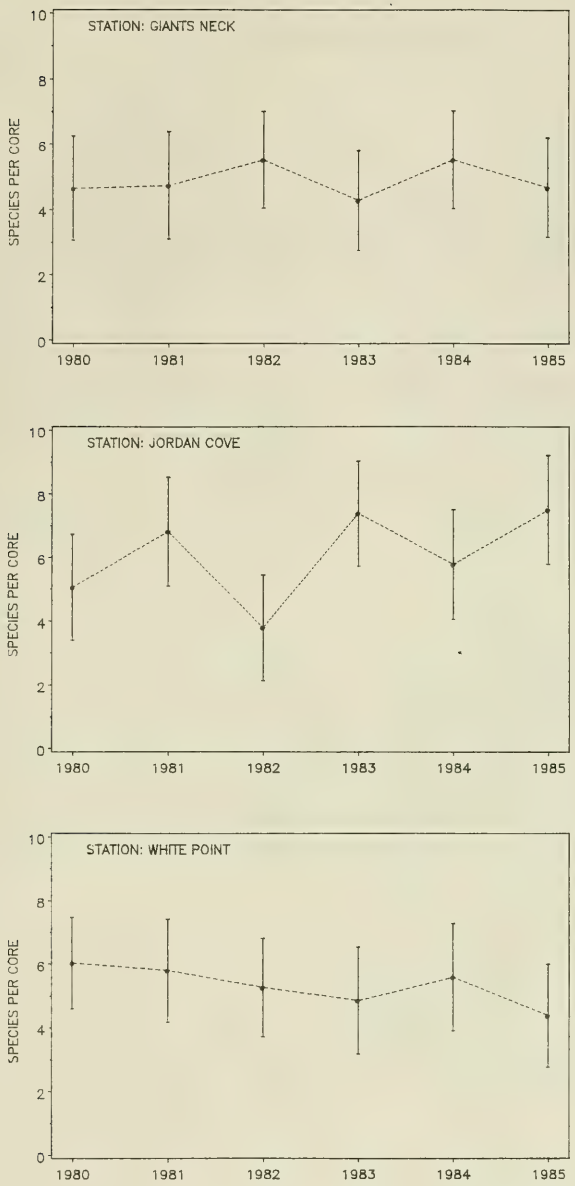

Figure 6. Annual mean number of species per core with two standard errors for intertidal infaunal communities sampled from September 1979 - June 1985. (Annual means were adjusted using analysis of covariance which included abiotic and climatic conditions as covariates.)

Table 4. Annual mean species diversity (H'), evenness (J), species number (S), total individual (± 1 standard error) collected at Millstone intertidal stations from September 1979 - June 1985.

Station	1980	1981	1982	1983	1984	1985
Giant Neck						
H'	1.9 ± 0.3	1.7 ± 0.2	1.5 ± 0.4	1.3 ± 0.2	2.2 ± 0.5	1.7 ± 0.4
J	0.7 ± 0.1	0.6 ± 0.1	0.4 ± 0.1	0.5 ± 0.1	0.6 ± 0.1	0.5 ± 0.1
S	9 ± 3	10 ± 1	13 ± 1	6 ± 1	13 ± 3	9 ± 2
N	322 ± 202	202 ± 73	368 ± 61	162 ± 87	220 ± 112	149 ± 54
Jordan Cove						
H'	1.5 ± 0.5	1.7 ± 0.5	1.6 ± 0.5	1.7 ± 0.5	1.5 ± 0.4	2.3 ± 0.1
J	0.4 ± 0.1	0.4 ± 0.1	0.5 ± 0.2	0.4 ± 0.1	0.5 ± 0.1	0.5 ± 0.1
S	12 ± 2	15 ± 5	9 ± 2	17 ± 4	12 ± 4	21 ± 3
N	267 ± 127	421 ± 160	309 ± 230	558 ± 162	581 ± 249	580 ± 237
White Point						
H'	2.2 ± 0.1	2.1 ± 0.2	1.2 ± 0.2	1.5 ± 0.1	1.4 ± 0.2	1.7 ± 0.4
J	0.6 ± 0.1	0.7 ± 0.1	0.4 ± 0.1	0.6 ± 0.1	0.4 ± 0.1	0.5 ± 0.1
S	12 ± 2	11 ± 3	10 ± 2	7 ± 2	10 ± 1	11 ± 2
N	193 ± 53	275 ± 101	337 ± 148	97 ± 39	181 ± 63	182 ± 58

Cluster Analysis

Cluster analysis, based on annual species counts since 1980 produced two major station-time groups that reflected differences between community structure at JC and those found at GN and WP (Fig. 7).

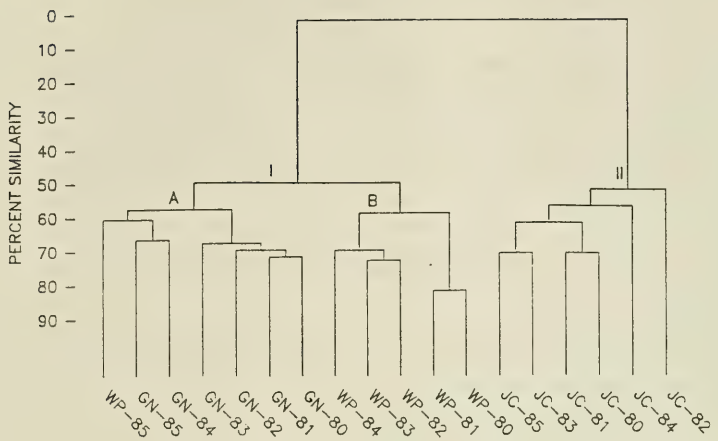

Figure 7. Dendrogram resulting from classification of annual infaunal collections at Millstone intertidal stations from September 1979 - June 1985.

The first major group of station/years (Group I) included all WP and GN collections and was futher subdivided into two subgroups. Group A contained GN years, and the 1985 WP collection; similarity among these station/years and their separation from other collections was due to shared high numbers of oligochaetes and *Haploscoloplos fragilis*. Subgroup B (remaining WP years) separated from other years because of very high densities of rhynchocoels, *Streptosyllis arenae* and *Paraonis fulgens*.

The JC collections (Group II) had relatively low similarity with GN and WP because of the high numbers of oligochaetes, *Scolecolepides viridis*, *Hediste diversicolor* and *Polydora ligni* found at JC. The relatively high similarity among most JC years reflected temporal constancy in the species composition at this station and the level of similarity among years was dependent on variations in the abundance of taxa rather than shifts in composition. For example, the subgroup containing 1981, 1982, 1983 and 1985 collections included very high numbers of oligochaetes, and high numbers of *Polydora ligni*, *Capitella* spp. and *Gammarus lawrencianus*. Collections from other years, had similar species composition, but densities of these species were considerably lower.

INTERTIDAL DISCUSSION

Intertidal infaunal beach communities were dominated by annelids, with arthropods, molluscs and rhynchocoels being locally abundant. The structure and composition of these communities contrasts sharply with haustoriid amphipod communities that have been reported as typical of more exposed sandy beaches along the coasts of North America (Croker et al. 1975; Croker 1977; Dexter 1969) and are more typical of annelid dominated communities reported from more sheltered estuarine areas (Withers and Thorpe 1978; Mauer and Aprill 1979; Tourtellotte and Dauer 1983).

Although the level of wave exposure within LIS is probably lower than ocean exposed beaches, wind and wave energy appears to be a major structuring force within the Millstone area. Throughout the monitoring program, all measures used to describe infaunal communities have consistently illustrated the spatial similarity of the GN and WP communities and their dissimilarity from that of JC. The shallow-sloping, clean sandy beaches of GN and WP, reflect their constant exposure to wind and wave-induced scour. Generally low numbers of species and individuals are found at these beaches and dominant components include *Haploscoloplos fragilis*, *Paraonis fulgens*, rhynchocoels, and *Streptosyllis arenae*, taxa which are typically found in sandy habitats (Dexter 1969; Whitlatch 1977; Maurer and Aprill 1979; Tourtellotte and Dauer 1983).

The clean medium sands consistently found at GN and WP contrasts sharply with the sedimentary environment at JC. The JC station is protected from constant wave scour, except during periods of strong southeast winds. Here the sediments are less uniform in size, exhibit larger seasonal shifts in size, contain higher amounts of silt/clay and are frequently covered by large amounts of algae or eelgrass. Infaunal abundance and numbers of species have been typically much higher at JC than at either GN or WP; differences in density are primarily caused by very high densities of oligochaetes. Other organisms, including molluscs and arthropods have been frequently more abundant at JC than at other stations. In intertidal sand habitats, higher numbers of these organisms generally occur in areas of increased shelter from wind-induced stress (Croker 1977, Maurer and Aprill 1979). The dominant components of the JC community include oligochaetes, *Hediste diversicolor*, *Capitella* spp. and *Polydora ligni*, species which can utilize fine organic matter as a food source (Sanders et al. 1962; Whitlatch 1977; Caspers 1980; Knott et al. 1983). High abundances of these taxa may also be enhanced by the detrital mat which

covers the JC beach throughout much of the year; this mat can significantly influence the numbers and types of species found inhabiting intertidal beaches (Soulsby et al. 1982).

Seasonal and year-to-year fluctuations in intertidal community abundance and composition were evident throughout the Millstone sampling program. However, an extensive evaluation of seasonal trends in species abundance are not appropriate because the sampling is performed only four times per year. A more important consideration, from the monitoring standpoint, is the removal of natural, temporal variation that can significantly reduce our ability to identify power plant related impacts on infaunal communities. Intertidal communites are strongly influenced by physical environmental conditions (Holland and Polgar 1976) and interannual variations in these conditions are expected to effect rather large changes in the overall abundance and composition of these communities. Attempts to model this variation at the community level, using abiotic factors as explanatory variables, removed 20-60% of the total variation in community density and 60-63% of the variation in numbers of species observed since 1980. After adjusting for the effects of the variation in these factors, both the average numbers of species and individuals have not significantly changed at any of our intertidal stations since 1980. Species compositional changes have also been relatively minor at our intertidal stations, and over the baseline period, were limited to rearrangements in the ranking of dominant species rather than additions or deletions.

INTERTIDAL CONCLUSIONS

Long-term studies of intertidal infaunal communites characterized areas communities within and beyond areas that will be potentially impacted by the Millstone facility. These data established spatial relationships among sampling stations, and investigated the extent and direction of temporal fluctuations in the abundance and numbers of species that have occurred from 1980 - 1985. During this period, no changes in intertidal communities could be attributed to operation of Millstone Units 1 and 2 or construction of Unit 3. Data presented here appear representative of natural, undisturbed intertidal communities and will be the basis for all future impact studies performed during 3-unit operating conditions.

SUBTIDAL RESULTS

Sedimentary Environment

Quarterly mean grain size and silt/clay content of subtidal sediments from June 1979 to March 1986 are plotted in Figure 8. Sediments at subtidal stations ranged from fine to coarse sands and contained up to 40% silt/clay. Bottom sediments at IN were composed of fine (0.125-0.25 mm) to medium (0.26-0.50 mm) sands. Since December 1983, seven of ten quarterly values were lower than those obtained prior to this period. At EF, sediments were composed of fine to medium sand. Coarse sands only predominated in March 1981. Grain size at this station was relatively stable at this station until June 1985; since then greater seasonal variations and a general increase in mean grain size have occurred. Sediments at GN were generally composed of medium sands, although over the sampling period they ranged from 0.14-0.76 mm. From March 1979 to March 1983, grain size at this station gradually increased and since then, relatively larger seasonal peaks in grain size have been evident during September. Sediments at JC were mostly medium sands, although coarse sands sometimes occurred during winter months (i.e., December and March). Of all stations, JC sediments were among the coarsest during the study.

Silt/clay content of subtidal sediments ranged from 1.3 to 44.1%. At GN, silt/clay ranged between 10% and 20% and has been relatively consistent since December 1980. At IN, silt/clay ranged from 5.5 to 44%, although 17 of the 28 values were between 5% and 10%. Since June 1983, IN silt/clay content increased to levels higher than those previously observed. Silt/clay values at JC ranged from 3 to 24%; however, 23 of 28 values were less than 10%. At EF, silt-clay content ranged from 2-14%, and 26 of 28 values were between 1% and 10%.

Sedimentary characteristics of all stations, except JC, have exhibited some temporal shifts during the study. Values for grain size at Giants Neck grain size have generally increased since 1982. This trend is believed attributable to the increased amounts of *Mytilus edulis* shell found in the sediments. This shell increases the overall weight of sediment retained by larger sieves during analysis and inflates grain size values. At IN, higher and more variable silt/clay values have been observed since June 1983, when construction activities (dredging and coffer dam removal) resulted in the deposition of fine material

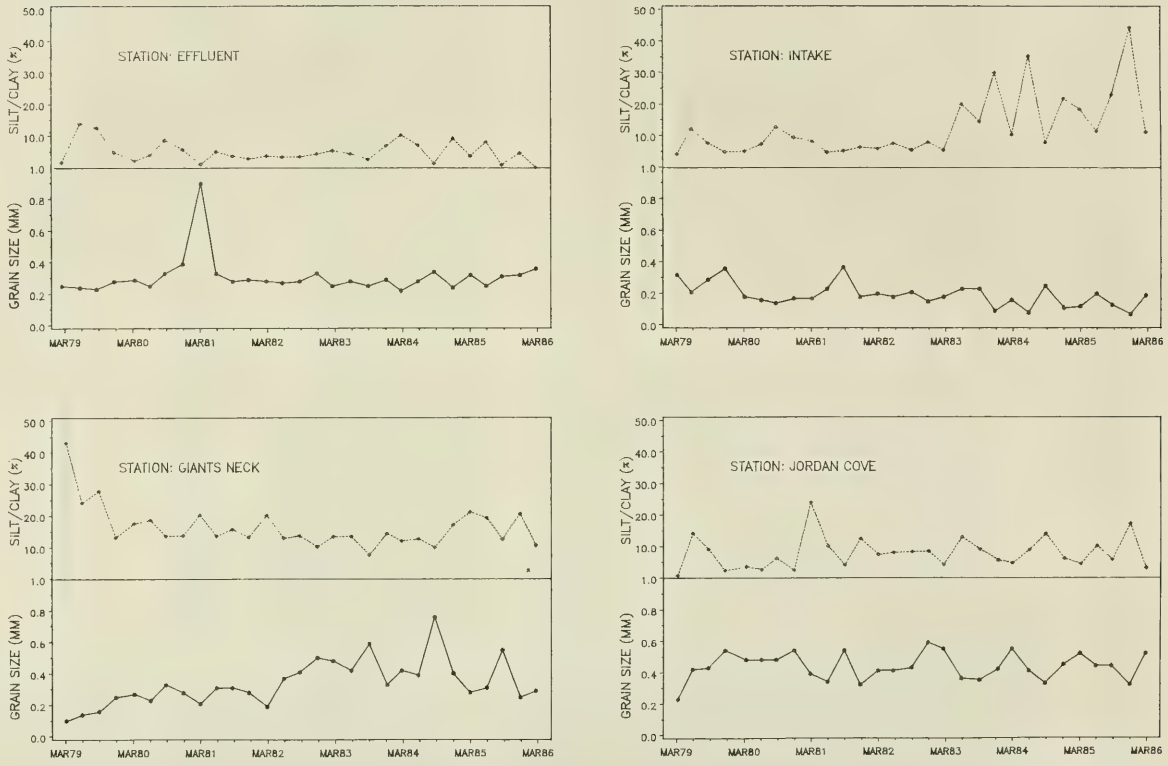

Figure 8. Quarterly mean grain size (mm) and silt/clay content (%) of sediments sampled at Millstone subtidal stations from March 1979 - March 1986.

at this station. Silt/clay content at EF has also become more variable following construction activities in the area of the Unit 3 discharge cut.

General Community Composition

The 960 samples collected from the four subtidal stations from September 1979 - June 1985 yielded a total of 359 taxa and 193,956 individuals. On an annual basis, polychaetes and oligochaetes were the most abundant organisms and collectively represented 77 to 91%, 88 to 92% and 58 to 96% of the individuals at EF, GN and JC, respectively (Table 5). At IN, these groups were also abundant (23 to 82%), although arthropods were generally more abundant than oligochaetes. Arthropods usually accounted for less than 10% of the individuals found at other stations. Molluscs and rhynchocoels were common at EF, but generally represented less than 10% of the total organisms collected.

Polychaetes frequently accounted for over half of all species collected (Table 5). On an annual basis, the number of polychaete species collected typically ranged from 60 to 75 at all but IN, where totals generally ranged between 40 and 50 species. Arthropods were the second most numerous species group at all stations (20 to 40 species) followed by molluscs (15 to 30 per year/station).

Quarterly total numbers of individuals and relative abundance for each major taxa collected at subtidal stations from September 1979 to March 1986 are presented in Appendix III. Polychaete species number and abundances were generally highest in either September or June. Although molluscs and arthropods contributed substantially to the subtidal communities, there were no consistent seasonal trends evidenced by these groups. In comparison to previous years, some changes in species composition were evident during 1986. Higher numbers of arthropod species were recorded in September and December samples at GN and JC than in previous years. At these stations, total species number in September collections was also high when compared to previous years. At IN, the numbers of species in 1986 continued to increase. This trend began in 1985 after construction activities (in 1984) eliminated most of the species in this area. In contrast, the numbers of species and individuals of all the major taxa at EF were generally lower in 1986 than in the previous two years.

Table 5. Number of species (S), number of individuals (N), and relative percent of total (%) for each major taxon collected at Millstone subtidal stations sampled from September 1979 - June 1985.

	1980			1981			1982			1983			1984			1985		
	S	N	%	S	N	%	S	N	%	S	N	%	S	N	%	S	N	%
Effluent																		
Polychaeta	62	5995	73	68	4596	53	65	2464	43	66	2714	40	68	7849	62	75	4432	39
Oligochaeta	-	1470	18	-	2595	30	-	2623	46	-	3181	47	-	2944	23	-	4496	39
Mollusca	29	370	5	27	478	6	25	140	2	26	144	2	34	915	7	31	934	8
Arthropod	41	318	4	37	799	9	32	411	7	45	636	9	40	762	6	40	1412	12
Rhynchocoela	-	75	1	-	202	2	-	92	2	-	47	1	-	168	1	-	245	2
'Others'	5	7	<1	4	3	<1	5	8	<1	5	35	1	5	10	<1	0	0	0
Totals	137	8235		136	8673		127	5738		142	6757		147	12648		146	11519	
Giants Neck																		
Polychaeta	72	8057	69	59	4262	58	68	7502	72	57	4332	70	74	9527	79	71	6415	66
Oligochaeta	-	2658	23	-	2198	30	-	2075	20	-	962	16	-	1528	13	-	2173	22
Mollusca	22	191	2	20	276	4	19	281	3	12	49	1	22	322	3	27	443	5
Arthropoda	31	686	6	33	535	7	42	558	5	32	761	12	35	565	5	35	637	7
Rhynchocoela	-	41	<1	-	28	<1	-	66	1	-	41	1	-	89	1	-	107	1
'Others'	4	23	<1	1	1	<1	4	9	<1	4	6	<1	4	8	<1	0	0	0
Totals	129	11656		113	7300		133	10491		105	6151		135	12039		133	9775	
Intake																		
Polychaeta	47	1330	66	41	1358	64	43	1218	52	44	866	47	38	1149	59	47	737	18
Oligochaeta	-	316	16	-	245	12	-	354	15	-	322	17	-	86	4	-	195	5
Mollusca	22	171	8	16	288	14	17	100	4	12	61	3	14	159	8	24	417	10
Arthropoda	28	196	10	25	222	11	31	667	28	19	593	32	19	545	28	28	2749	67
Rhynchocoela	-	14	1	-	7	<1	-	9	<1	-	10	1	-	15	1	-	33	1
'Others'	0	0	0	2	3	<1	1	1	<1	0	0	0	0	0	0	0	0	0
Totals	97	2027		84	2120		92	2348		75	1852		71	1954		99	4131	
Jordan Cove																		
Polychaeta	68	6412	43	68	4283	49	57	2576	35	52	5504	51	66	13357	37	70	6943	57
Oligochaeta	-	7811	53	-	3669	42	-	4570	59	-	4621	43	-	3877	21	-	195	5
Mollusca	26	348	2	32	539	6	14	109	1	15	204	2	28	717	4	27	760	6
Arthropoda	25	242	2	25	285	3	20	297	4	28	352	3	30	256	2	31	2412	2
Rhynchocoela	-	74	1	-	56	1	-	77	1	-	66	1	-	125	1	-	78	1
'Others'	1	3	<1	4	4	<1	3	3	<1	2	5	<1	6	8	<1	0	0	0
Totals	120	14890		129	8836		94	7732		97	10752		130	18342		128	12105	

Community Abundance

Mean quarterly abundance (no./core) of subtidal communities from March 1979 - March 1986 ranged from 60 to 393 at EF, 107 to 360 at GN, 9 to 303 at IN and 106 to 633 at JC, (values are exponentials of those presented in Fig. 9). At EF, GN and JC, abundance was generally highest in June or September with declines evident during colder months of December and March. This seasonal pattern was evident at IN until June 1984 after which peaks in density occurred in September and December.

Regression analysis using abiotic factors as explanatory variables (i.e., covariates) removed 46%, 48%, 55% and 57% of the temporal variation in community abundance at EF, GN, IN, and JC, respectively. After removing abiotic sources of variation, no significant long-term increases or decreases in annual mean abundance (for 1980 - 1985 only) were evident. In fact, overall abundance at all stations was relatively stable over the sampling period (Fig. 10). Inter-annual differences did occur; however, and paired-t tests of adjusted annual mean revealed significant $\alpha < 0.05$ differences at all stations. Except at IN, these differences were generally due to the very high abundances that occurred in past years. For example, the high density at JC in 1984, was significantly different from 1981 to 1983 and 1985. Similarly, the high 1984 density at EF was significantly different from all previous years. In contrast, the low density at IN in 1984 was significantly different from every other year except 1981. Lower density during 1981, at GN also resulted in significant differences with 1980 and 1984. In addition, the low 1983 density was significantly different from densities in 1980, 1982, 1984 and 1985.

Numbers of Species

The mean quarterly species numbers (no./core) at subtidal stations ranged from 5-23 at IN, 19-40 at JC, 12-40 at GN and 13-46 at EF (Fig. 11). Species numbers were highest at EF and lowest at IN. Seasonal patterns generally followed those of density (higher in June or September at EF, GN and JC and December at IN).

Multiple regressions of species numbers accounted for 75% (EF), 63% (JC), 61% (GN) and 47% (IN) of the temporal variation observed since 1980. At EF the model included a significant increasing trend which persisted after removal of quantifiable sources of variation (Fig. 12). Results of paired t-tests showed that significant inter-annual differences occurred at all stations and were frequently in years in

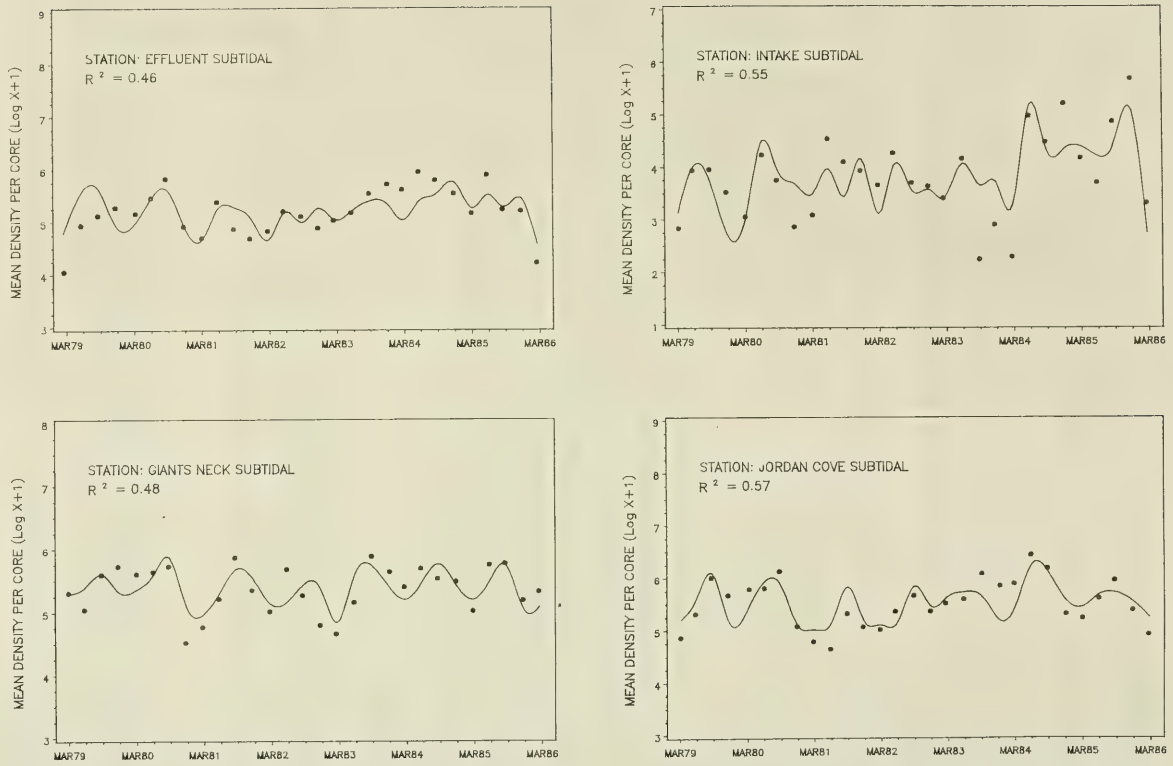

Figure 9. Quarterly mean numbers of individuals per core and multiple regression predictions for Millstone subtidal infaunal communities sampled from March 1979 - March 1986.

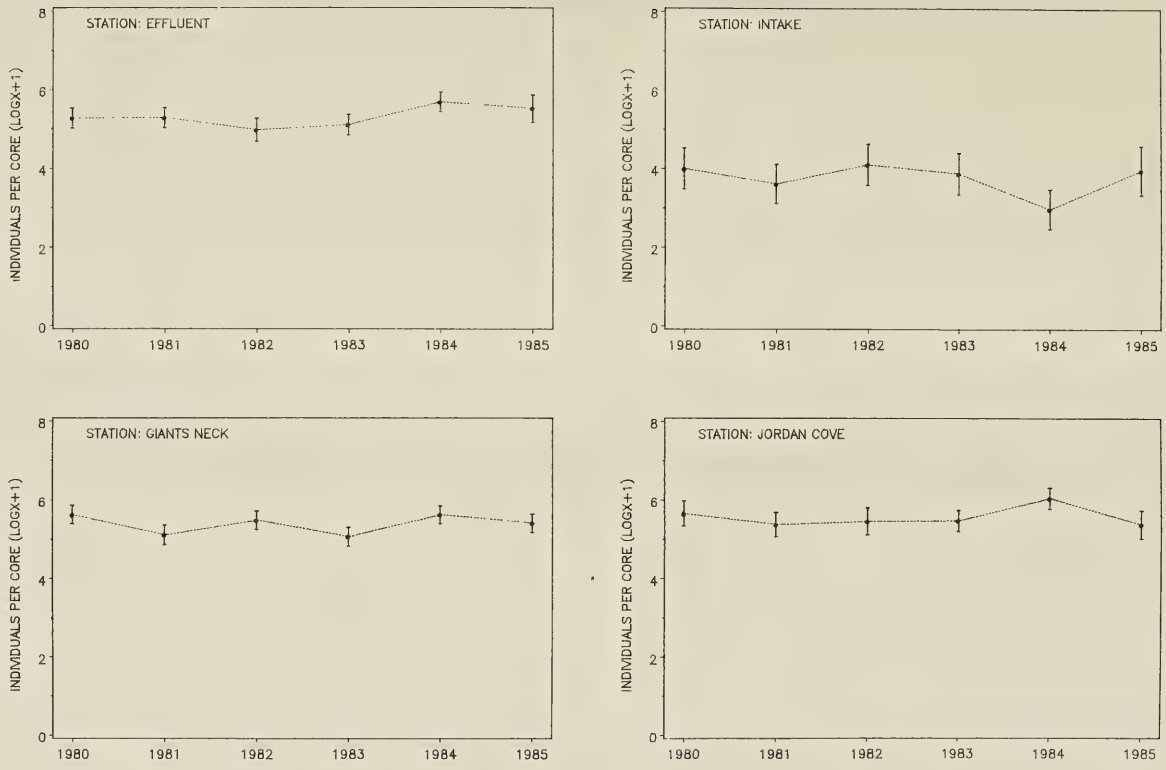

Figure 10. Annual mean abundance per core with two standard errors for subtidal infaunal communities sampled from September 1979 - June 1985. (Annual means were adjusted using analysis of covariance which included abiotic and climatic conditions as covariates.)

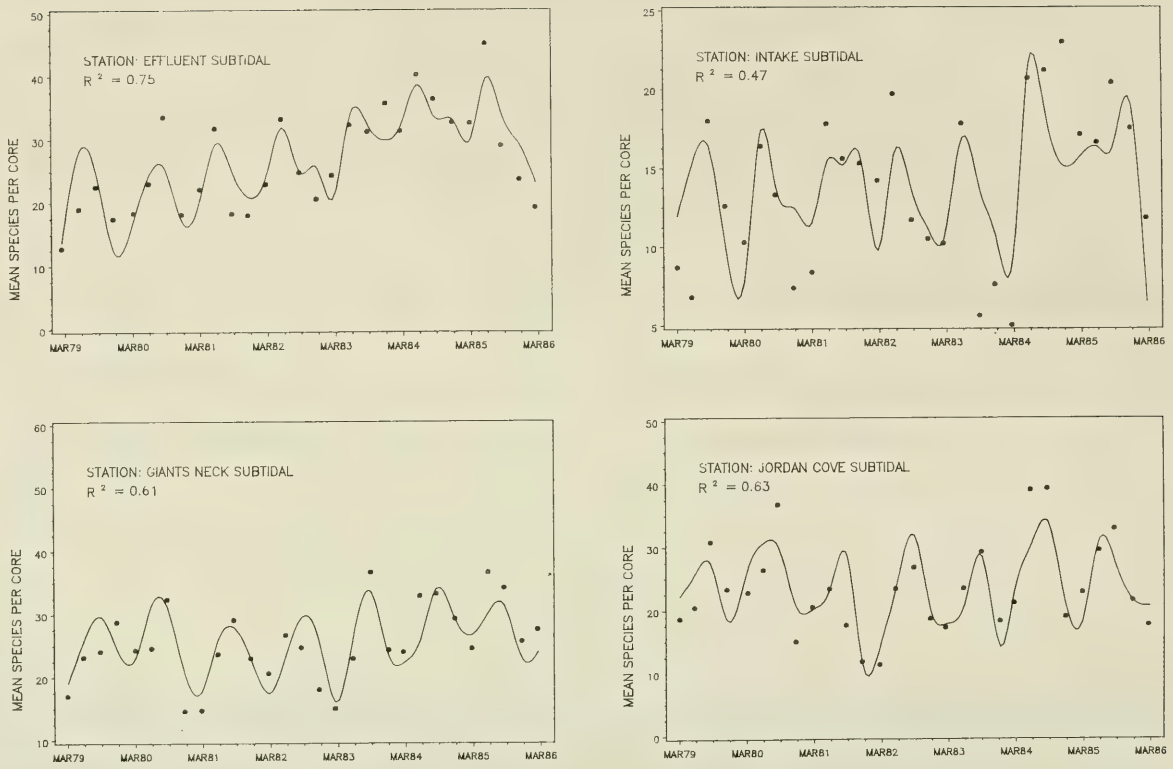

Figure 11. Quarterly mean numbers of species per core and multiple regression predictions for Millstone subtidal infaunal communities sampled from March 1979 - March 1986.

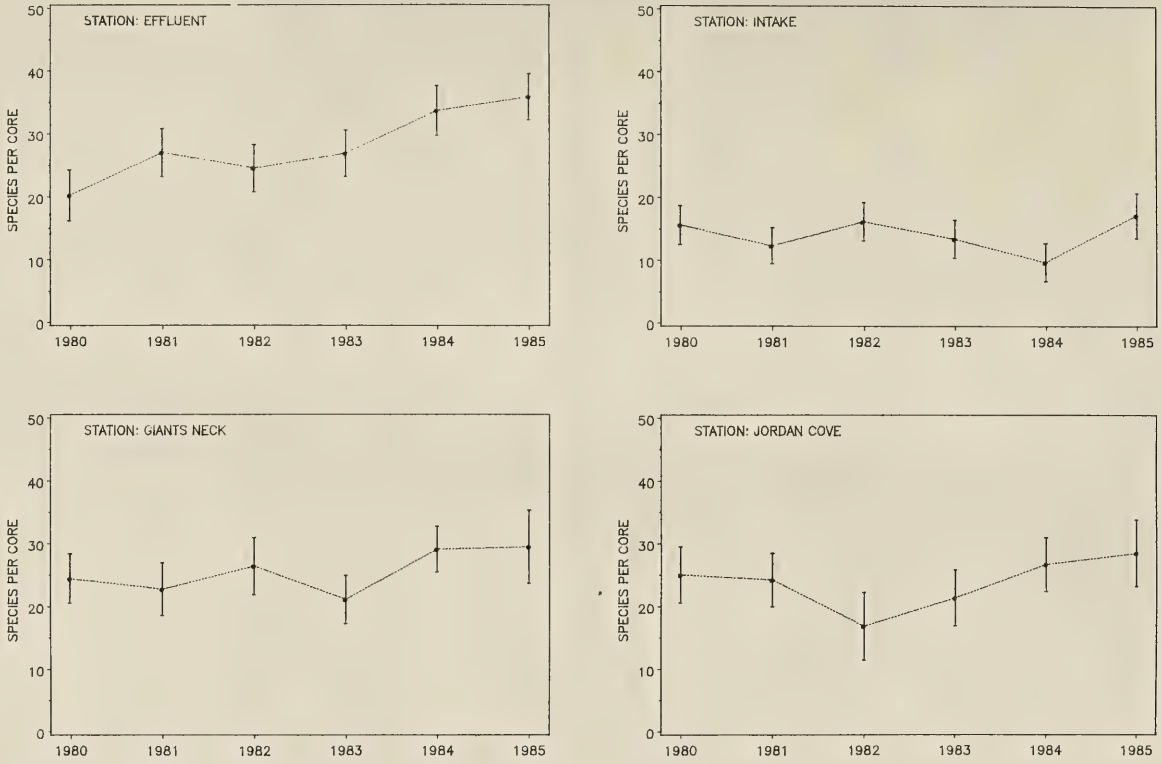

Figure 12. Annual mean number of species per core with two standard errors for subtidal infaunal communities sampled from September 1979 - June 1985. (Annual means were adjusted using analysis of covariance which included abiotic and climatic conditions as covariates.)

which very low numbers of species were collected. The annual mean at IN in 1984 was significantly lower than in 1980 and 1982. At JC, the mean number of species in 1982 was significantly lower than those in 1980, 1981, 1984 and 1985, and the low mean values obtained in 1981 and 1983, were lower than 1984 and 1985. Only at EF did this pattern differ; 1984 and 1985 values were significantly higher than all previous years.

Dominance

Since 1980, a composite list of all taxa present among the ten numerically abundant forms in any sampling year, included 18, 19, 21 and 24 taxa at GN, JC, EF and IN, respectively (Table 6). Among these, eight taxa were dominant at all stations: *Tharyx acutus*, oligochaetes, *Aricidea catherinae*, *Capitella* spp., *Gammarus lawrencianus*, *Prionospio steestrupi*, *Leptocheirus pinguis* and *Mediomastus ambiseta*.

Over all stations and years, oligochaetes (as a group) were the most consistently dominant taxon, accounting for 17 to 51% (EF), 13 to 31% (GN), 4 to 19% (IN) and 21 to 61% (JC) of the total individuals collected and thus had the highest BIV's (93.1-99.1%). *Aricidea catherinae* was also consistently found among the dominants at GN, IN and JC, where it ranked second in terms of the six-year BIV and generally accounted for over 10% of all the individuals collected. Other taxa among the more consistently abundant forms (i.e., BIV $>80\%$) were *Polycirrus eximius* and *Protodorvillea gaspeensis* at EF, *Tharyx* spp. at GN and *Tharyx acutus* at JC.

Of all communities, that of IN was the most spatially dissimilar; 29% of the taxa included among the ten dominants collected over the last six years, were amphipods. In addition, species type was highly variable and only oligochaetes had a BIV $>$ 80%. Many other dominant taxa at this station exhibited strong temporal fluctuations in abundance. For example, the relative abundance of *Tellina agilis* ranged from 1-15%, *Mediomastus ambiseta* from 1-20%, *Ampelisca verrilli* from $<$ 1-24% and *Polydora ligni* from 0-19%. No other sampling station has exhibited this degree of variability.

Table 6. Percent contribution and Biological Index Value (BIV) of the ten numerically most abundant taxa at Millstone subtidal stations in each year, September 1979 - June 1985.

Station	1980	1981	1982	1983	1984	1985	BIV
Effluent							
Oligochaeta	17	31	47	51	24	39	98.4
Polycirrus eximius	17	7	6	6	32	8	92.9
Protodorvillea gaspeensis	1	7	7	5	2	3	81.7
Aricidea catherinae	12	1	4	7	2	1	69.4
Tharyx acutus	34	21	1	3	1	1	65.5
Tellina agilis	1	2	1	1	4	4	65.1
Rhynchocoela	1	2	2	1	1	2	59.5
Tharyx spp.	<1	1	5	1	1	4	57.9
Exogone hebes	1	1	2	2	1	1	54.4
Eumidia sanguinea	1	3	2	<1	1	1	53.2
Capitella spp.	1	1	1	1	1	1	47.2
Mediomastus ambiseta	<1	9	<1	<1	8	5	46.0
Polydora caulleryi	<1	<1	<1	4	1	1	43.7
Lumbrineris tenuis	<1	<1	1	<1	2	2	41.3
Caulleriella spp.	<1	<1	2	1	1	1	38.9
Pagurus acadianus	0	<1	1	1	2	<1	34.9
Leptocheirus pinusuis	<1	<1	<1	1	1	2	32.5
Ampelisca verrilli	<1	<1	<1	2	<1	1	31.3
Microphthalmus aberrans	1	<1	1	<1	<1	<1	31.0
Prionospio steenstrupi	1	<1	2	<1	<1	<1	29.0
Gammarus lawrencianus	<1	2	1	<1	<1	<1	26.2
Giants Neck							
Oligochaeta	25	31	21	17	13	23	94.4
Aricidea catherinae	19	16	35	26	13	14	94.4
Tharyx spp.	9	12	10	19	16	16	90.7
Tharyx acutus	14	11	5	4	1	4	75.9
Protodorvillea gaspeensis	2	3	3	3	4	3	68.5
Lumbrineris tenuis	1	1	3	4	2	3	63.2
Mediomastus ambiseta	9	1	1	1	25	7	60.2
Polycirrus eximius	5	2	2	1	3	3	58.3
Phoxoxephalus holbolli	3	3	1	1	1	2	51.2
Exogone dispar	1	1	1	1	3	5	48.6
Polydora caulleryi	<1	<1	<1	3	3	2	42.6
Prionospio steenstrupi	1	1	2	1	1	<1	35.2
Capitella spp.	1	2	1	<1	<1	1	34.7
Gammarus lawerencianus	0	2	1	3	<1	<1	33.8
Lumbrineris impatiens	1	1	2	0	0	<1	32.4
Eumida sanguinea	<1	1	<1	<1	1	1	32.4
Polydora quadrilobata	<1	<1	<1	1	2	<1	25.5
Leptochierus pinguis	<1	<1	<1	2	<1	<1	17.6

Table 6 (con't)

Intake

Oligochaeta	16	18	15	19	4	5	93.1
Aricidea catherinae	8	15	12	13	1	4	79.2
Phyllodoce mucosa	1	2	<1	<1	0	<1	70.5
Capitella spp.	13	7	3	3	1	<1	67.4
Tellina agilis	5	15	2	2	1	3	67.0
Exogone hebes	5	4	6	7	2	1	64.9
Tharyx acutus	9	9	2	2	<1	2	62.8
Mediomastus ambiseta	5	1	1	1	20	5	60.1
Spiophanes bombyx	4	1	6	2	1	1	57.6
Unciola serrata	<1	1	4	2	2	4	57.3
Nucula proxima	<1	1	1	1	5	5	52.1
Leptocheirus pinguis	<1	<1	1	2	10	17	50.0
Prionospio steenstrupi	2	1	2	1	5	<1	49.0
Ampelisca abdita	<1	<1	1	2	7	26	49.0
Protodorvillea gaspeensis	1	2	3	1	<1	<1	46.2
Ampelisca verrilli	3	2	8	24	<1	5	45.5
Ampelisca vadorum	<1	<1	2	1	2	8	45.5
Clymenella torquata	1	1	8	1	0	0	41.7
Polydora quadrilobata	<1	<1	2	4	1	<1	39.9
Pygospio elegans	1	1	<1	0	3	0	38.9
Polydora ligni	<1	1	0	<1	19	1	36.1
Gammarus lawrencianus	<1	3	6	1	<1	<1	29.5
Sabellaria vulgaris	6	<1	<1	0	0	<1	24.3
Crangon septemspinosus	<1	0	<1	<1	4	<1	22.6

Jordan Cove

Oligochaeta	51	41	61	43	21	34	99.1
Aricidea catherinae	8	21	17	25	10	11	92.1
Tharyx spp.	1	2	3	1	2	5	80.7
Polycirrus eximius	4	3	11	3	5	8	79.8
Mediomastus ambiseta	19	2	1	2	43	11	77.6
Parapionosyllis longicirrata	1	1	1	<1	<1	<1	67.5
Lumbrineris impatiens	2	1	1	<1	<1	0	55.3
Lumbrineris tenuis	2	6	2	3	4	8	52.2
Tellina agilis	<1	3	<1	<1	2	2	50.0
Polydora caulleryi	<1	<1	1	8	2	1	46.7
Rhynchocoela	<1	1	1	1	1	1	43.9
Eumida sanguinea	1	2	<1	<1	<1	1	40.4
Tharyx acutus	2	2	<1	2	1	1	36.8
Capitella spp.	1	1	1	2	1	3	35.5
Pholoe minuta	<1	<1	1	1	<1	<1	34.2
Exogone hebes	<1	1	<1	<1	<1	1	31.1
Prionospio steenstrupi	<1	<1	1	<1	1	<1	28.9
Leptocheirus pinguis	<1	<1	<1	<1	<1	<1	26.8
Gammarus lawrencianus	<1	<1	1	<1	<1	<1	19.3

Species Diversity

Mean annual species diversity (H') ranged from 2.6 to 4.6 (Table 7). Low diversity indices (e.g., 1980 at EF and 1984 at JC) were seen when short-term pulses in species abundance caused the evenness component of diversity to decline. High H' values reflected changes in the numbers of individuals or species, but not in the equality of the distribution of individuals among species.

Table 7. Annual mean species diversity (H'), evenness (J), species number (S) and total individual collected at Millstone subtidal stations from September 1979 - June 1985.

Station	1980	1981	1982	1983	1984	1985
Effluent						
H'	2.7 ± 0.2	3.9 ± 0.3	4.4 ± 0.2	4.6 ± 0.1	3.6 ± 0.2	4.8 ± 0.1
J	0.4 ± 0.1	0.7 ± 0.1	0.7 ± 0.1	0.7 ± 0.1	0.6 ± 0.1	0.7 ± 0.1
S	63 ± 6	66 ± 13	63 ± 9	75 ± 8	84 ± 4	86 ± 5
N	1583 ± 29	1324 ± 364	689 ± 144	809 ± 88	2333 ± 211	1667 ± 228
Giant Neck						
H'	3.6 ± 0.1	3.6 ± 0.2	3.4 ± 0.1	3.4 ± 0.1	3.5 ± 0.2	4.2 ± 0.1
J	0.6 ± 0.1	0.6 ± 0.1	0.6 ± 0.1	0.6 ± 0.1	0.6 ± 0.1	0.6 ± 0.1
S	68 ± 4	57 ± 10	69 ± 6	53 ± 5	73 ± 8	82 ± 4
N	2080 ± 46	1177 ± 394	1975 ± 456	1230 ± 175	2549 ± 217	1824 ± 281
Intake						
H'	4.1 ± 0.1	3.8 ± 0.1	3.9 ± 0.1	3.4 ± 0.3	3.4 ± 0.2	3.8 ± 0.3
J	0.8 ± 0.1	0.7 ± 0.1	0.7 ± 0.1	0.7 ± 0.1	0.7 ± 0.1	0.7 ± 0.1
S	44 ± 4	37 ± 6	45 ± 3	37 ± 3	30 ± 8	51 ± 3
N	389 ± 797	301 ± 102	474 ± 71	369 ± 81	445 ± 335	907 ± 293
Jordan Cove						
H'	3.6 ± 0.2	3.7 ± 0.4	3.0 ± 0.4	3.0 ± 0.2	2.6 ± 0.2	3.8 ± 0.2
J	0.7 ± 0.1	0.6 ± 0.1	0.6 ± 0.1	0.5 ± 0.1	0.4 ± 0.1	0.6 ± 0.1
S	66 ± 1	66 ± 9	44 ± 6	55 ± 4	67 ± 9	72 ± 9
N	1694 ± 480	1202 ± 4	724 ± 145	1477 ± 174	3561 ± 523	1949 ± 711

During the baseline period, evenness (J) ranged from 0.4 to 0.8, numbers of species from 30 to 86 and individuals from 301 to 3,561. Values for J were generally highest at IN and most consistent at GN (0.6 in every year); EF and GN had the most species and IN generally the fewest. The numbers of species collected at EF, GN and JC increased from 1982 through 1985 when high numbers of species were collected at all stations. The IN station had consistently had lowest mean numbers of individuals in all sampling years.

Quarterly values of species diversity from September 1979 to March 1986, for subtidal stations are presented in Appendix IV. At all subtidal stations, H′, S and J were generally highest in the September or June. At EF, all diversity values in the three 1986 (September and December 1985 and March 1986) were lower than similar periods for the last two years. Values of H′ over the same period at GN were slightly lower than 1985, but very comparable to previous years. Because of the high density of *Ampelisca* spp. in the December IN collections, H′ was lower than all previous observations, otherwise, values were similar to previous years. At JC, all species diversity index components in 1986, were similar to previous years.

Cluster Analysis

Cluster analysis, performed to examine spatial and temporal similarities among subtidal collections, clearly illustrated the spatially distinct nature of the infaunal community found at IN (Group I), and the strong spatial similarity among collections from EF, GN and JC (Group II) (Fig. 13). Within the IN group (I), collections made in 1985 and 1984 separated from previous years due to a species compositional shift. In each of the last two years, large increases in the numbers of ampeliscid arthropods have occurred, together with a marked reduction in annelid number (particularly oligochaetes).

Group II included all other collections and the primary cluster was subdivided into station groups (A, B, C). Within each group, collections made during 1985 and 1984 linked at highest similarity reflecting the area-wide abundance of oligochaetes and *Mediomastus ambiseta*. In addition, species that have been traditionally dominant at subtidal stations were found in unusually high abundance in these years. For example at EF, high densities of *Lumbrineris tenuis*, *Ampelisca verrilli* and *Leptocheirus pinguis* occurred in each of the last two years, while at GN and JC, *Tharyx* spp. and *Polycirrus eximius* were very abundant.

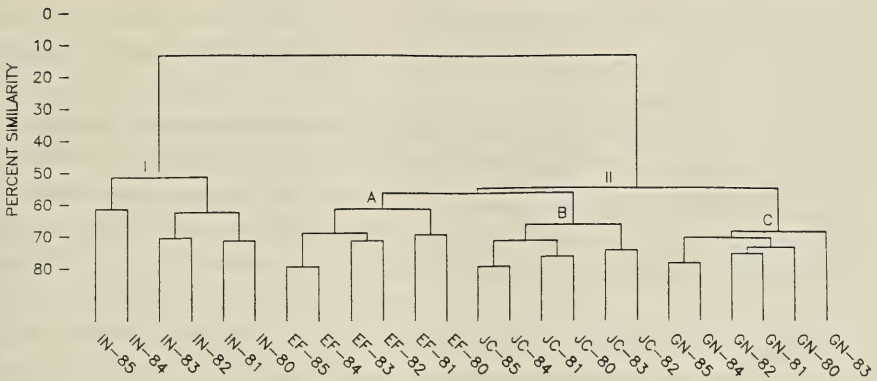

Figure 13. Dendrogram resulting from classification of annual infaunal collections at Millstone subtidal stations from September 1979 - June 1985.

SUBTIDAL DISCUSSION

The subtidal benthic monitoring program characterized infaunal communities inhabiting three areas potentially impacted by power plant operations (IN, EF and JC) and an unimpacted reference area (GN). Temporal and spatial changes in species abundance and overall community composition were evident at IN and EF and appear related to power plant to construction and related dredging. Differences also occurred at GN and JC, but given the limited area influenced by two-unit operations, changes are probably reflecting natural events. The following section is intended to summarize the spatial and temporal changes observed over the monitoring period and assess whether these changes were related to power plant operation.

During the baseline period, infaunal community abundance and species composition at IN reflected the unique nature of this station and exhibited impacts associated with construction and dredging activities which have occurred in the area. Prior to construction, a variety of amphipods were present at this tidally scoured station and infaunal communities were more typical of those found in offshore areas

where bottom currents are higher (Biernbaum 1979). After construction, the infaunal community responded to bottom disturbances in a highly predictable manner. As in other areas subjected to disturbance, (i.e., dredging) the initial decrease in species numbers and abundance was followed by the rapid invasion of opportunistic species (e.g., *Polydora ligni*, *Capitella Mediomastus ambiseta*), (Grassle and Grassle 1974; McCall 1977; Swartz et al. 1980; Flint and Younk 1983; Nichols 1985). Following these species, ampeliscid amphipods, more traditionally found at the IN station, settled and became very abundant. These species can stabilize the sediment surface (Blumer et al. 1970; Sanders et al. 1972) and thus allow recolonization and recruitment of other species to the area.

Power plant related impacts at EF, even during the period immediately following construction activities, were less dramatic than those at IN. Levels of silt/clay were only slightly elevated and apparently enhanced colonization of species that rely on fine material as a food source. In 1985, densities of deposit feeding species such as oligochaetes, *Tharyx* spp., *Tellina agilis*, *Lumbrineris tenuis* and *Polycirrus eximius* remained higher than most previous years (except 1984). Although these species were present at this station in the past, their relative abundances were much lower before construction. The EF community also exhibited a significant increase in the average number of species. Although relatively high numbers of species have been collected at this station, the most dramatic increase occurred after construction. This increase probably reflected the influx of deposit-feeding species, which utilized the higher silt/clay content found after construction began. However, since the opening of the second discharge cut, silt/clay content decreased and the numbers of amphipod and mollusc species increased in the three quarters during the 1986 sampling period. Despite these changes, the structure of the EF community has exhibited a higher degree of year-to-year similarity than that at IN.

Generally, the GN and JC communities have been more similar to each other in terms of species composition, abundance and diversity. Species comprising these communities and their abundances are more similar to other studies of near-shore areas (e.g., Watling 1975) than those of IN. During the baseline period, an area-wide increase and decline was seen in the abundance of *Mediomastus ambiseta* at GN and JC. This species was the most abundant organism at both stations during 1984, and accounted for 25% (GN) and 43% (JC) of the total organisms collected. In 1985, the relative abundance of *Mediomastus ambiseta* declined to 7% and 12% at GN and JC, respectively (lower densities of this taxon were also evident at EF and IN in 1985). The decreased abundance of *Mediomastus ambiseta* at

JC and GN was accompanied by increases in more traditionally dominant members of these communities (e.g., *Lumbrineris tenuis*, *Aricidea catherinae*, *Polycirrus eximius*)

Temporal shifts in community composition and abundance have occurred throughout the study at all subtidal stations. This variation often reflects the interaction of biotic and abiotic factors (Coull 1985; Holland 1985; Nichols 1985) and temporal shifts in numbers of species, abundance, diversity and species composition are characteristic of shallow-water benthic communities (Green 1969; Eagle 1975; McCall 1977; Watling 1975; Holland and Mountford 1977; Rachor and Gerlach 1978; Loi and Wilson 1979). Even though subtidal abundances in the 1986, were lower than in recent years, regression models, which included natural abiotic factors as variables, accounted for the lower values; thus the observed differences were not due to plant operation. Shifts in infaunal species abundance probably reflected variations in recruitment success and mortality, processes which are frequently linked to predation and changes in the local physical and chemical environments (Nichols 1985; Gallagher et al. 1983). The short-term rearrangements in the ranking of a few of the dominant species apparently reflect the effect of these natural phenomena (Watling 1975; Flint and Younk 1983; Nichols and Thompson 1985).

SUBTIDAL CONCLUSIONS

The Millstone subtidal benthic monitoring program detected some power plant related impacts at stations in the immediate vicinity of the discharge (EF) and intake (IN) structures. Observed changes occurred prior to 1985, and were attributed to Unit 3 construction activities and not to operation of Millstone Units 1 and 2. Recent observation of community parameters at IN indicate the area is recovering from the disturbance. The opening of the second discharge cut may have caused recent changes in sedimentary and community parameters observed at EF. The widespread nature of infaunal community changes at GN and JC suggest a response to naturally occurring events and are not related to operation or construction of the Millstone facility. Data obtained during this study period will enable us to distinguish any future impacts due to 3-unit operation of the Millstone facility from those that occur naturally.

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Appendix I. Quarterly numbers of species (S), number of individuals and relative percent (%) for each major taxon collected at Millstone Point intertidal stations sampled for September 1979 - March 1986.

	September			December			March			June		
	S	N	%	S	N	%	S	N	%	S	N	%
Giants Neck												
1980												
Polychaeta	16	883	92	3	7	9	2	10	2	6	407	81
Oligochaeta	-	51	5	-	0	0	-	27	5	-	34	6
Mollusca	2	6	<1	1	1	1	1	1	<1	0	0	0
Arthropoda	1	1	<1	2	2	3	3	7	1	0	0	0
Rhynchocoela	-	20	2	-	68	87	-	459	91	-	64	13
Totals	19	961		6	78		6	504		6	505	
1981												
Polychaeta	11	497	86	7	158	57	4	35	19	10	447	80
Oligochaeta	-	24	4	-	2	<1	-	10	6	-	26	5
Mollusca	1	1	<1	0	0	0	0	0	0	2	2	<1
Arthropoda	0	0	0	2	8	3	2	7	4	0	0	0
Rhynchocoela	-	53	9	-	110	40	-	129	71	-	82	15
Totals	12	575		9	278		6	181		12	557	
1982												
Polychaeta	11	255	73	9	372	88	8	389	95	11	518	80
Oligochaeta	-	37	11	-	13	3	-	7	2	-	57	9
Mollusca	0	0	0	0	0	0	0	0	0	1	6	1
Arthropoda	1	1	<1	5	8	2	1	6	1	4	15	2
Rhynchocoela	-	57	16	-	31	7	-	6	1	-	55	8
Totals	12	350		14	424		9	408		16	651	
1983												
Polychaeta	6	494	75	4	68	29	4	24	38	6	135	76
Oligochaeta	-	51	8	-	5	2	-	25	40	-	14	8
Mollusca	0	0	0	0	0	0	0	0	0	0	0	0
Arthropoda	0	4	<1	2	7	3	2	2	3	1	3	1
Rhynchocoela	-	106	16	-	156	66	-	12	19	-	26	15
Totals	6	655		6	236		6	63		7	178	
1984												
Polychaeta	13	341	52	3	19	20	11	56	30	9	501	87
Oligochaeta	-	295	45	-	22	23	-	90	48	-	51	9
Mollusca	3	12	2	1	1	1	1	2	1	0	0	0
Arthropoda	1	2	<1	1	3	3	6	16	9	0	0	0
Rhynchocoela	-	11	2	-	51	53	-	22	12	-	25	4
Totals	17	661		5	96		18	186		9	577	
1985												
Polychaeta	13	227	65	6	248	68	4	25	22	7	113	16
Oligochaeta	-	49	14	-	8	2	-	40	36	-	529	73
Mollusca	0	0	0	0	0	0	0	0	0	1	1	<1
Arthropoda	0	0	0	3	5	1	1	3	2	3	4	<1
Rhynchocoela	-	74	21	-	106	29	-	44	40	-	7	-
TOTALS	13	350		9	367		5	109		11	720	
1986												
Polychaeta	2	239	42	3	48	81	9	12	23	-	-	-
Oligochaeta	48	-	0	0	-	14	27	-	-	-		
Mollusca	1	1	<1	0	0	0	0	0	0	-	-	-
Arthropoda	0	0	0	0	0	0	1	1	2	-	-	-
Rhynchocoela	-	-	58	10	-	11	19	-	25	48	-	-
Totals	3	568		3	59		10	52		-	-	-

Appendix 1 (cont.)

	September			December			March			June		
	S	N	%	S	N	%	S	N	%	S	N	%
Jordan Cove												
1980												
Polychaeta	11	359	13	4	45	2	8	16	9	4	597	15
Oligochaeta	-	2447	87	-	2066	97	-	94	54	-	3307	84
Mollusca	4	12	<1	2	2	<1	3	55	32	1	2	<1
Arthropoda	3	7	<1	1	4	<1	0	0	0	5	10	<1
Rhynchocoela	-	2	<1	-	7	<1	-	9	5	-	23	<1
Totals	18	2827		7	2124		11	174		10	3939	
1981												
Polychaeta	13	2416	42	12	214	15	4	26	2	5	185	34
Oligochaeta	-	3256	57	-	1191	83	-	948	70	-	324	60
Mollusca	3	23	<1	4	21	1	4	379	28	2	8	1
Arthropoda	12	49	<1	1	10	<1	0	0	0	1	2	<1
Rhynchocoela	-	6	<1	-	4	<1	-	8	<1	-	18	3
Totals	28	5750		17	1440		8	1361		8	537	
1982												
Polychaeta	8	64	19	6	30	9	2	139	15	3	479	17
Oligochaeta	-	268	78	-	287	88	-	788	84	-	1868	65
Mollusca	1	1	<1	1	1	<1	0	0	0	3	3	<1
Arthropoda	3	7	2	1	2	<1	1	4	<1	8	532	18
Rhynchocoela	-	3	<1	-	8	2	-	4	<1	0	0	0
Totals	12	343		8	328		3	935		14	2882	
1983												
Polychaeta	14	932	60	13	423	15	7	214	18	7	477	6
Oligochaeta	-	527	34	-	2340	81	-	988	82	-	7197	94
Mollusca	3	7	<1	2	3	<1	1	3	<1	3	3	<1
Arthropoda	9	75	5	5	113	4	0	0	0	3	17	<1
Rhynchocoela	-	1	<1	-	0	0	-	0	0	-	0	0
Totals	26	1542		20	2879		8	1205		13	7694	
1984												
Polychaeta	9	863	52	5	112	47	2	121	40	3	1081	63
Oligochaeta	-	763	46	-	42	18	-	175	59	-	605	35
Mollusca	2	5	<1	8	71	30	0	0	0	1	19	1
Arthropoda	10	40	2	3	10	4	1	1	<1	4	9	<1
Rhynchocoela	-	1	<1	-	1	<1	-	2	<1	-	4	<1
Totals	21	1672		17	236		3	299		8	1718	
1985												
Polychaeta	13	1110	35	10	206	8	8	101	4	12	716	8
Oligochaeta	-	2000	63	-	2306	89	-	2751	96	-	7955	91
Mollusca	4	45	1	2	3	<1	3	13	<1	2	6	<1
Arthropoda	5	13	<1	8	81	3	1	1	<1	15	80	1
Rhynchocoela	-	7	<1	-	3	<1	-	3	<1	-	4	<1
Totals	22	3175		20	2599		12	2869		29	8761	
1986												
Polychaeta	14	967	56	6	233	14	9	440	27	-	-	-
Oligochaeta	-	719	41	-	1344	80	-	1210	73	-	-	-
Mollusca	2	29	2	1	2	<1	2	2	<1	-	-	-
Arthropoda	7	22	1	6	14	<1	1	1	<1	-	-	-
Rhynchocoela	-	2	<1	-	77	5	-	2	<1	-	-	-
Totals	23	1739		13	1670		12	1655		-	-	-

Appendix I (cont.)

	September			December			March			June		
	S	N	%	S	N	%	S	N	%	S	N	%
White Point												
1980												
Polychaeta	16	370	89	9	150	57	9	66	17	10	266	60
Oligochaeta	-	11	3	-	3	1	-	3	<1	-	42	9
Mollusca	1	1	<1	3	3	1	0	0	0	1	1	1
Arthropoda	0	0	0	0	0	0	0	0	0	0	0	0
Rhynchocoela	-	35	8	-	106	40	-	317	82	-	136	31
Totals	17	417		12	262		9	386		11	445	
1981												
Polychaeta	15	599	67	8	272	53	4	5	4	10	362	47
Oligochaeta	-	168	19	-	9	2	-	21	16	-	75	10
Mollusca	0	0	0	0	0	0	0	0	0	1	3	<1
Arthropoda	1	1	<1	0	0	0	0	0	0	1	4	<1
Rhynchocoela	-	127	14	-	229	45	-	106	80	-	321	42
Totals	16	895		8	510		4	132		12	765	
1982												
Polychaeta	6	295	61	9	175	67	6	276	85	13	802	54
Oligochaeta	-	175	36	-	24	9	-	19	6	-	451	30
Mollusca	0	0	0	0	0	0	0	0	0	1	1	<1
Arthropoda	2	3	<1	0	0	0	0	0	0	1	3	<1
Rhynchocoela	-	14	3	-	63	24	-	28	9	-	232	16
Totals	8	487		9	262		6	323		15	1489	
1983												
Polychaeta	8	178	68	8	184	42	2	4	25	5	62	34
Oligochaeta	-	53	20	-	30	7	-	9	56	-	15	8
Mollusca	0	0	0	0	0	0	0	0	0	1	3	2
Arthropoda	1	1	<1	2	2	<1	1	1	6	1	1	<1
Rhynchocoela	-	28	11	-	222	51	-	2	13	-	104	56
Total	9	260		10	438		3	16		7	185	
1984												
Polychaeta	8	373	72	6	104	47	6	59	27	11	210	28
Oligochaeta	-	123	24	-	1	<1	-	16	7	-	137	18
Mollusca	0	0	0	2	2	<1	1	1	<1	1	1	<1
Arthropoda	1	3	<1	1	1	<1	1	5	2	2	2	<1
Rhynchocoela	-	18	3	-	114	51	-	135	63	-	412	54
Totals	9	516		9	222		8	216		14	762	
1985												
Polychaeta	12	264	32	8	116	79	4	50	57	6	284	68
Oligochaeta	-	424	51	-	14	10	-	9	10	-	13	3
Mollusca	1	1	<1	2	2	1	2	2	2	3	13	3
Arthropoda	2	2	<1	0	0	0	0	0	0	5	8	2
Rhynchocoela	-	133	16	-	15	10	-	26	30	-	99	24
Totals	15	824		10	147		6	87		14	417	
1986												
Polychaeta	5	168	55	2	5	20	4	40	80	-	-	-
Oligochaeta	-	26	8	-	4	16	-	3	6	-	-	-
Mollusca	0	0	0	1	2	8	0	0	0	-	-	-
Arthropoda	3	4	1	1	1	4	1	3	6	-	-	-
Rhynchocoela	-	109	36	-	13	52	-	4	8	-	-	-
Totals	8	307		4	25		5	50				

Appendix II. Quarterly species diversity (H'), evenness (J) and number of species (S) for each Millstone intertidal station sampled from September 1979 - March 1986.

	Giants Neck			Jordan Cove			White Point		
	H'	S	J	H'	S	J	H'	S	J
1980									
Sept.79	2.30	18	0.55	2.64	18	0.63	2.00	16	0.50
Dec. 79	2.42	6	0.94	1.65	7	0.59	2.37	12	0.66
Mar. 80	1.95	6	0.75	1.58	11	0.46	1.93	9	0.61
June 80	1.07	6	0.41	0.30	10	0.09	2.55	11	0.74
1981									
Sept.80	1.56	12	0.44	2.78	28	0.58	2.45	16	0.61
Dec. 80	1.63	9	0.51	2.25	17	0.55	1.22	7	0.44
Mar. 81	2.35	6	0.91	0.62	8	0.21	1.92	4	0.96
June 81	1.30	12	0.36	1.26	8	0.42	2.44	14	0.64
1982									
Sept.81	2.49	13	0.67	2.63	12	0.73	1.22	8	0.37
Dec. 81	0.83	14	0.22	2.06	8	0.69	1.64	9	0.52
Mar. 82	0.88	10	0.27	0.17	3	0.11	0.65	6	0.25
June 82	1.77	16	0.44	1.43	14	0.37	1.33	15	0.34
1983									
Sept.82	1.62	6	0.63	2.62	26	0.56	1.58	9	0.50
Dec. 82	1.12	6	0.43	2.59	21	0.59	1.32	10	0.40
Mar. 83	1.55	6	0.60	0.94	8	0.31	1.52	3	0.96
June 83	0.98	7	0.35	0.58	13	0.16	1.44	7	0.51
1984									
Sept.83	2.22	18	0.53	1.55	21	0.35	0.85	9	0.27
Dec. 83	2.11	5	0.91	2.54	17	0.62	1.53	9	0.48
Mar. 84	3.45	18	0.83	1.04	3	0.66	1.90	8	0.63
June 84	0.97	9	0.31	0.75	8	0.25	1.45	14	0.38
1985									
Sept.84	2.43	12	0.68	2.32	22	0.52	2.71	16	0.68
Dec. 84	0.76	9	0.24	2.12	20	0.49	1.68	10	0.51
Mar. 85	1.30	5	0.56	2.15	12	0.60	1.70	6	0.66
June 85	2.20	11	0.64	2.54	29	0.52	0.85	13	0.23
1986									
Sept.85	0.21	3	0.13	2.27	23	0.50	1.49	8	0.50
Dec. 85	0.39	3	0.25	1.21	13	0.33	1.75	4	0.88
Mar. 86	3.32	10	0.98	1.76	12	0.49	1.03	5	0.44

Appendix III. Quarterly number of species (S), number of individuals (N) and relative percent of the total (%) for each major taxon collected at Millstone subtidal stations from September 1979 to March 1986.

	September			December			March			June		
	S	N	%	S	N	%	S	N	%	S	N	%
Effluent												
1980												
Polychaeta	41	1438	81	28	1531	73	28	2166	88	35	1847	64
Oligochaeta	-	101	6	-	393	19	-	132	5	-	844	29
Mollusca	11	60	3	9	93	4	17	111	5	12	106	4
Arthropoda	24	162	9	14	65	3	15	33	1	14	58	2
Rhynchocoela	-	12	1	-	11	<1	-	24	1	-	28	1
Totals	76	1773		51	2093		60	2466		61	2883	
1981												
Polychaeta	49	2126	56	24	839	58	35	724	62	40	1012	43
Oligochaeta	-	1186	31	-	437	30	-	234	20	-	738	31
Mollusca	17	227	6	6	28	2	8	21	2	17	202	9
Arthropoda	30	156	4	8	111	8	10	152	13	19	380	16
Rhynchocoela	-	122	3	-	28	2	-	34	3	-	18	<1
Totals	96	3817		38	1443		53	1165		76	2358	
1982												
Polychaeta	30	523	37	29	411	36	37	951	56	49	2398	72
Oligochaeta	-	756	54	-	623	55	-	609	36	-	635	19
Mollusca	12	64	6	6	20	2	13	32	2	14	24	<1
Arthropoda	11	40	3	9	55	5	15	80	5	21	236	7
Rhynchocoela	-	17	1	-	19	2	-	15	1	-	41	1
Totals	43	1400		44	1128		65	1687		84	3334	
1983												
Polychaeta	32	639	35	38	489	33	35	841	49	53	1217	55
Oligochaeta	-	854	47	-	862	59	-	703	41	-	762	34
Mollusca	12	33	2	10	32	2	13	29	2	13	50	2
Arthropoda	21	276	15	22	67	5	18	130	8	28	163	7
Rhynchocoela	-	4	<1	-	10	1	-	3	<1	-	29	1
Totals	65	1806		70	1460		66	1706		94	2221	
1984												
Polychaeta	39	1573	60	45	1784	56	43	2007	72	54	2470	62
Oligochaeta	-	636	24	-	994	31	-	462	17	-	852	21
Mollusca	15	173	7	23	248	8	15	201	7	15	293	7
Arthropoda	21	232	9	25	154	5	15	112	4	21	264	7
Rhynchocoela	-	19	<1	-	21	<1	-	11	<1	-	117	3
Totals	75	2633		93	3201		73	2793		90	3996	
1985												
Polychaeta	40	1260	39	42	1086	40	40	978	50	57	1235	33
Oligochaeta	-	1288	40	-	1127	42	-	647	33	-	1434	38
Mollusca	20	267	8	21	275	10	16	160	8	16	232	6
Arthropoda	26	312	10	18	185	7	18	143	7	28	772	21
Rhynchocoela	0	123	3	-	27	1	-	13	<1	-	82	2
Totals	86	3250		81	2700		74	1941		101	3755	
1986												
Polychaeta	31	626	33	33	776	43	29	1401	79	-	-	-
Oligochaeta	-	867	46	-	910	61	-	262	15	-	-	-
Mollusca	16	149	8	18	102	6	10	74	4	-	-	-
Arthropoda	21	234	12	16	137	8	11	31	2	-	-	-
Rhynchocoela	-	22	1	-	20	1	-	14	<1	-	-	-
Totals	69	1898		67	1788		50	1782		-	-	-

Appendix III (cont.)

	September			December			March			June		
	S	N	%	S	N	%	S	N	%	S	N	%
Jordan Cove												
1980												
Polychaeta	44	2978	67	32	381	14	39	1320	37	40	1132	32
Oligochaeta	–	1237	28	–	2203	81	–	2131	60	–	2240	63
Mollusca	12	114	3	15	79	3	13	84	2	15	71	2
Arthropoda	12	74	2	14	44	2	14	44	1	13	80	2
Rhynchocoela	–	12	<1	–	18	<1	–	10	<1	–	34	1
	Totals	68	4415		61	2725		66	3589		68	3557
1981												
Polychaeta	46	2280	47	26	50	4	36	663	53	42	673	63
Oligochaeta	–	2114	44	–	910	84	–	437	35	–	208	20
Mollusca	26	232	5	12	84	8	16	110	9	13	113	11
Arthropoda	17	150	3	7	30	3	10	39	3	13	66	6
Rhynchocoela	–	29	<1	–	14	1	–	8	<1	–	5	<1
	Totals	89	4805		45	1088		62	1257		68	1065
1982												
Polychaeta	29	969	44	22	264	18	25	463	30	37	781	34
Oligochaeta	–	1127	51	–	1171	78	–	1030	67	–	1242	54
Mollusca	11	68	3	7	12	<1	3	6	<1	6	22	1
Arthropoda	8	19	1	6	36	2	4	25	2	15	217	9
Rhynchocoela	–	8	<1	–	23	2	–	15	<1	–	31	1
Totals	48	2191		35	1506		32	1540		58	2293	
1983												
Polychaeta	36	1504	50	27	114	8	33	1670	64	35	1407	49
Oligochaeta	–	1269	42	–	1211	85	–	856	33	–	1285	44
Mollusca	8	65	2	8	34	2	5	24	1	11	81	3
Arthropoda	19	156	5	15	50	3	6	42	2	11	104	4
Rhynchocoela	–	11	<1	–	23	2	–	20	<1	–	12	<1
Totals	63	3005		50	1432		44	2612		57	2889	
1984												
Polychaeta	44	3782	74	31	65	5	31	2962	80	50	4587	71
Oligochaeta	–	920	20	–	1010	81	–	624	17	–	1323	21
Mollusca	20	204	4	12	139	11	12	85	2	16	289	4
Arthropoda	10	61	1	9	16	1	6	11	<1	22	168	3
Rhynchocoela	–	10	<1	–	19	2	–	38	1	–	58	1
Totals	74	4577		52	1249		49	3720		88	6425	
1985												
Polychaeta	52	3628	73	33	261	15	36	1057	53	38	1447	50
Oligochaeta	–	831	17	–	1328	77	–	732	37	–	1192	41
Mollusca	22	334	7	12	105	6	19	166	8	16	155	5
Arthropoda	18	137	3	11	19	1	7	12	1	24	73	3
Rhynchocoela	–	39	<1	–	12	1	–	17	1	–	10	<1
Totals	92	4969		56	1725		62	1984		78	2877	
1986												
Polychaeta	42	2081	51	35	463	32	20	908	64	–	–	–
Oligochaeta	–	1490	37	–	840	57	–	386		–	–	–
Mollusca	26	395	10	12	69	5	12	37	3	–	–	–
Arthropoda	24	85	2	16	66	4	15	76	5	–	–	–
Rhynchocoela	–	26	<1	–	23	2	–	9	1	–	–	–
Totals	92	4077		63	1461		47	1416		–	–	–

Appendix III. (cont.)

		September			December			March			June		
		S	N	%	S	N	%	S	N	%	S	N	%
Giants Neck													
1980													
Polychaeta	43	2046	74	42	1998	63	38	2166	77	38	1847	64	
Oligochaeta	–	542	20	–	957	27	–	491	17	–	768	27	
Mollusca	9	52	2	14	58	2	13	45	2	12	36	1	
Arthropoda	11	113	4	21	238	8	17	108	4	10	227	8	
Rhynchocoela	–	6	<1	–	10	<1	–	10	<1	–	15	<1	
	Totals	63	2759		71	3161		68	2820		59	2893	
1981													
Polychaeta	49	2059	65	21	459	47	28	724	60	37	1020	53	
Oligochaeta	–	698	22	–	433	44	–	443	37	–	624	32	
Mollusca	14	187	6	8	28	3	4	14	1	11	47	2	
Arthropoda	22	210	7	12	60	6	10	24	2	13	241	12	
Rhynchocoela	–	12	<1	–	1	<1	–	6	<1	–	9	<1	
	Totals	85	3166		41	981		42	1211		60	1941	
1982													
Polychaeta	41	2738	74	43	1415	63	31	951	62	38	2398	79	
Oligochaeta	–	597	16	–	649	29	–	508	33	–	321	11	
Mollusca	16	168	5	11	49	2	6	10	<1	10	54	2	
Arthropoda	24	162	4	16	90	4	14	58	4	21	248	8	
Rhynchocoela	–	12	<1	0	26	1	–	9	<1	–	19	<1	
Totals	81	3677		70	2229		51	1536		63	3040		
1983													
Polychaeta	40	1368	69	25	906	72	24	841	76	36	1217	67	
Oligochaeta	–	384	19	–	232	18	–	221	20	–	125	7	
Mollusca	6	15	1	7	17	1	4	6	<1	4	11	<1	
Arthropoda	18	200	10	14	97	7	12	32	3	16	432	24	
Rhynchocoela	–	6	<1	–	9	1	–	7	<1	–	19	1	
Totals	64	1973		46	1261		40	1107		63	1804		
1984													
Polychaeta	52	2794	75	37	2256	78	39	2007	85	44	2470	81	
Oligochaeta	–	491	13	–	472	16	–	283	12	–	282	9	
Mollusca	19	133	4	9	67	2	11	43	2	11	79	3	
Arthropoda	25	304	8	13	78	3	13	28	1	14	155	5	
Rhynchocoela	–	11	<1	–	16	<1	–	7	<1	–	55	2	
Totals	96	3733		59	2889		63	2368		75	3041		
1985													
Polychaeta	44	1691	66	41	1607	66	41	978	62	55	2139	66	
Oligochaeta	–	505	20	–	615	25	–	396	25	–	657	20	
Mollusca	7	140	5	14	84	3	18	114	7	17	105	3	
Arthropoda	20	179	7	21	99	4	12	74	5	21	285	9	
Rhynchocoela	–	33	1	–	28	1	–	15	1	–	31	1	
Totals	71	2548		76	2433		71	1577		78	3217		
1986													
Polychaeta	46	2113	64	35	1085	58	40	1401	64	–	–	–	
Oligochaeta	–	545	17	–	426	23	–	524	24	–	–	–	
Mollusca	20	154	5	15	54	3	15	68	3	–	–	–	
Arthropoda	25	470	14	22	285	15	28	189	9	–	–	–	
Rhynchocoela	–	11	<1	–	13	<1	–	15	<1	–	–	–	
Totals	91	3293		72	1863		83	2197		–	–	–	

Appendix III. (cont.)

		September			December			March			June		
		S	N	%	S	N	%	S	N	%	S	N	%
Intake													
1980													
Polychaeta	35	368	64	23	381	76	17	113	52	20	468	64	
Oligochaeta	–	81	14	–	66	13	–	46	21	–	123	17	
Mollusca	8	46	8	9	28	6	7	28	13	9	69	9	
Arthropoda	12	75	13	11	22	4	11	31	14	16	68	9	
Rhynchocoela	–	5	1	–	3	1	–	1	<1	–	5	1	
	Totals	55	575		43	500		35	219		45	733	
1981													
Polychaeta	25	289	61	11	50	28	11	80	36	27	939	75	
Oligochaeta	–	85	18	–	55	31	–	46	21	–	59	5	
Mollusca	9	48	10	6	44	25	10	73	33	5	123	10	
Arthropoda	15	51	11	7	27	15	7	23	10	14	121	10	
Rhynchocoela	–	2	<1	–	0	0	–	0	0	–	5	<1	
	Totals	49	475		24	176		28	222		46	1247	
1982													
Polychaeta	25	463	73	21	264	46	22	235	60	27	256	34	
Oligochaeta	–	68	11	–	153	27	–	42	11	–	91	12	
Mollusca	11	46	7	8	24	4	7	10	3	2	20	3	
Arthropoda	19	58	9	14	124	22	11	107	27	16	378	3	
Rhynchocoela	–	2	<1	–	4	<1	–	1	<1	<1	2	<1	
Totals	55	637		43	569		40	394		45	747		
1983													
Polychaeta	23	133	33	22	114	29	20	120	38	29	499	70	
Oligochaeta	–	109	25	–	105	26	–	34	11	–	74	10	
Mollusca	4	12	3	5	15	4	2	9	3	6	25	4	
Arthropoda	11	175	41	10	166	42	7	145	46	9	107	15	
Rhynchocoela	–	1	<1	–	0	0	–	5	2	–	4	<1	
Totals	38	430		37	400		29	313		44	709		
1984													
Polychaeta	10	31	34	11	65	31	15	39	37	30	1014	66	
Oligochaeta	–	23	26	–	2	1	–	9	9	–	52	3	
Mollusca	4	11	12	7	22	10	6	54	51	9	72	5	
Arthropoda	7	24	26	7	114	54	0	0	0	14	407	26	
Rhynchocoela	–	1	1	–	8	4	–	3	3	–	3	<1	
Totals	21	90		25	211		21	105		53	1548		
1985													
Polychaeta	26	210	22	24	261	13	17	124	15	25	142	33	
Oligochaeta	–	17	2	–	70	4	–	69	9	–	39	9	
Mollusca	12	158	17	13	159	8	12	82	10	7	18	4	
Arthropoda	17	544	58	17	1447	74	16	529	65	19	229	53	
Rhynchocoela	–	15	1	–	11	<1	–	4	<1	–	3	<1	
Totals	55	944		54	1948		45	808		51	431		
1986													
Polychaeta	23	585	42	25	463	15	25	195	63	–	–	–	
Oligochaeta	–	45	3	–	22	<1	–	38	12	–	–	–	
Mollusca	12	331	24	6	201	7	5	16	5	–	–	–	
Arthropoda	19	429	31	15	2364	77	14	57	19	–	–	–	
Rhynchocoela	–	6	<1	–	14	<1	–	2	<1	–	–	–	
Totals	54	1396		46	3064		44	308		–	–	–	

Appendix IV. Quarterly species diversity (H'), evenness (J) and number of species (S) for each Millstone subtidal station sampled from September 1979 - March 1986

	Effluent			Giants Neck			Intake			Jordan Cove		
	H'	S	J	H'	S	J	H'	S	J	H'	S	J
1980												
Sept.79	3.09	79	0.49	3.40	66	0.56	4.42	54	0.77	3.03	67	0.50
Dec. 79	2.29	51	0.40	3.82	78	0.61	3.77	43	0.69	3.75	62	0.63
Mar. 80	2.49	60	0.42	3.30	68	0.54	4.19	34	0.82	3.58	66	0.59
June 80	2.29	61	0.48	3.72	61	0.63	4.02	43	0.73	4.14	67	0.68
1981												
Sept.80	3.48	97	0.53	3.95	84	0.62	3.79	50	0.67	4.02	90	0.62
Dec. 80	3.45	38	0.66	3.46	41	0.65	3.68	25	0.79	2.64	46	0.48
Mar. 81	4.00	53	0.70	3.19	42	0.59	3.50	28	0.73	3.68	61	0.62
June 81	4.72	76	0.75	3.97	60	0.67	4.14	46	0.75	4.41	68	0.72
1982												
Sept.81	3.86	53	0.67	3.19	82	0.50	3.56	55	0.62	2.87	49	0.51
Dec. 81	4.31	45	0.78	3.46	71	0.56	4.11	43	0.76	2.51	36	0.48
Mar. 82	4.54	67	0.75	3.54	51	0.62	3.92	39	0.74	2.57	32	0.51
June 82	4.89	85	0.76	3.23	71	0.53	4.11	44	0.75	4.11	58	0.70
1983												
Sept.82	4.29	64	0.72	3.51	65	0.58	3.43	38	0.65	3.33	65	0.55
Dec. 82	4.73	71	0.77	3.35	47	0.60	3.04	36	0.59	2.85	51	0.50
Mar. 83	4.39	67	0.72	3.10	41	0.58	3.24	29	0.67	2.64	45	0.48
June 83	4.84	97	0.73	3.43	57	0.59	3.96	43	0.73	3.20	57	0.55
1984												
Sept.83	3.42	78	0.54	3.95	96	0.36	3.79	20	0.88	2.92	77	0.47
Dec. 83	3.87	92	0.59	3.03	61	0.28	3.18	25	0.69	2.10	53	0.37
Mar. 84	3.26	74	0.52	3.26	64	0.34	3.09	21	0.70	2.37	50	0.42
June 84	3.92	90	0.60	3.85	70	0.30	3.45	53	0.60	3.01	89	0.46
1985												
Sept.84	4.51	87	0.70	4.25	84	0.66	4.16	56	0.72	3.63	96	0.55
Dec. 84	4.61	84	0.72	4.00	78	0.64	3.15	54	0.55	3.65	55	0.63
Mar. 85	4.75	75	0.76	4.15	72	0.67	3.54	44	0.65	3.82	61	0.64
June 85	5.15	100	0.77	4.21	93	0.64	4.44	51	0.78	4.29	77	0.68
1986												
Sept.85	4.66	70	0.76	3.93	92	0.60	3.45	54	0.60	3.75	94	0.57
Dec. 85	3.92	70	0.64	3.81	73	0.62	2.04	46	0.37	3.46	66	0.57
Mar. 86	4.41	50	0.78	3.99	83	0.63	3.94	43	0.73	3.40	46	0.62

Contents

LOBSTER POPULATION DYNAMICS 1
INTRODUCTION 1
MATERIALS AND METHODS 2
RESULTS AND DISCUSSION 6
Abundance and Catch Per Unit Effort 6
Population Characteristics 12
Size Frequencies 12
Sex Ratios 15
Reproductive Activities 16
Molting and Growth 19
Claw Loss 25
Tagging Program 26
Movement 27
Entrainment 30
Impingement 33
SUMMARY 35
CONCLUSION 36
REFERENCES CITED 37

LOBSTER POPULATION DYNAMICS

INTRODUCTION

The American lobster, *Homarus americanus*, is the most valuable commercially harvested species in Long Island Sound (LIS). Annual landings since 1977 have ranged between 600,000 and 2,000,000 pounds with a value ranging between 1.3 and 6.2 million dollars. The landings for New London county, which include Millstone Point, were between 28% and 45% of the total LIS catch from 1977 to 1985 (Blake and Smith 1984; CT DEP personal communications). Exploitation rates in LIS are high and over 90% of marketable lobsters are newly recruited from the sublegal size class (Smith 1977; Keser et al. 1983). Therefore the strength of the legal catch is highly dependent on the number of lobsters in the prerecruit size class (one molt from legal size).

The lobster monitoring program at the Millstone Nuclear Power Station (MNPS) was designed to assess the impacts of plant operations by evaluating year-to-year, seasonal, and between station changes in selected population characteristics such as catch per unit effort, size frèquencies, growth rates, sex ratios, female size at sexual maturity, characteristics of egg-bearing females and lobster movements. Lobster larvae entrainment studies are also conducted to assess impacts on the larval stage of lobsters. The results of all these studies are often compared to other studies conducted throughout the range of the American lobster.

Potential effects of MNPS operations on the lobster population are impingement of lobsters on the intake traveling screens, entrainment of larvae through the cooling water systems, and thermal effects of the discharge. These power plant impacts may reduce survival of lobster larvae and juveniles or alter the behavior of adults which may result in a decline in the local inshore fishery.

The purpose of this report is to summarize the results of the lobster population studies conducted during 2-unit operation from 1975 to 1985. These results will be compared to data collected under 3-unit operating conditions to assess the possible impacts on the local lobster population associated with MNPS operations.

MATERIALS AND METHODS

The collection of information on the abundance of lobsters in the Millstone area began in February 1969, when the numbers of legal, short, and berried lobsters were recorded using catch records from a local commercial lobsterman (Table 1).

Table 1. Summary of lobster population sampling methodology from 1969 to 1985.

Year	Sample Period	Sample Method	Sample Stations
1969	June-Dec	Commercial Catch Records	Millstone, Spindle Area, Seaside, White Rock, Bartlett Reef
1970-73	Feb, May, July[a] Sept, Dec	Commercial Catch Records SCUBA Surveys Battelle Pots (6 wood/Station)	Seaside, Fox Island, Bartlett Reef
1973	Oct	Monthly Surveys of Artificial and Natural Habitats (SCUBA)	Effluent, Intake, Twotree, Giants Neck, Bay Point
1974	Apr-Dec	"	"
1975	Jan-Dec	"	"
1975	Sept-Dec	Battelle Pots (80 wood)[b]	Effluent, Jordan Cove, Twotree, Intake
1976-77	Jan-Dec	"	"
1978	Jan-Aug	NUSCo Pots (60 wood)	Jordan Cove, Intake, Twotree
1978	Aug-Dec	NUSCo Pots (30 wood and 30 wire)	"
1979-80	May-Oct	"	"
1981-85	May-Oct	NUSCo Pots (60 wire)	"

a Single Pots set out for 1 week and checked daily.
b Pots hauled three times per week, weather permitting, from 1975 through 1985.

During the period 1970-73 lobster pot sampling was conducted daily for one week in February, May, July, September and December. Six wood pots were set for a week around Fox Island (at the fringe of the Unit 1 plume) and, as controls, six were set around Seaside Point (1.5 miles from the plant); in May 1973 the Bartlett's Reef station was added. These pots were checked daily during the week that the Rocky Shore surveys took place. In addition to the pot sampling, several SCUBA surveys were made annually to assess the abundance of lobsters in these areas.

During the summer of 1973, artificial habitats consisting of an array of 36 concrete blocks (16 x 24 x 10 in), with three burrows each, were installed to provide additional habitat at four sites: outside of the quarry-cut, in Jordan Cove, near the intake structures, and adjacent to Bartlett's Reef. Four natural

lobster areas were chosen for observation in conjunction with the artificial habitats: located south of the discharge, between Twotree Island and Bartlett's Reef, near Bay Point, and south of Giants Neck. Each artificial habitat and natural area was inspected monthly using SCUBA. These areas were monitored through December 1975, when artificial habitat monitoring was replaced by a more intense tagging program using pots to collect lobsters.

Beginning in September 1975, pot trawls consisting of five double entry wood pots (3-5 cm lath spacings) strung along a 50-75 m line bouyed at both ends were used to collect lobsters. Four pot trawls were placed at Jordan Cove, Intake, Effluent and Twotree Island (Fig. 1).

Figure 1. Location of the Millstone Nuclear Power Station (MNPS) and the three lobster sampling stations (●).

Pots were checked three times each week, rebaited with flounder carcasses and reset in the same area. Lobsters > 55 mm carapace length (CL) were banded to restrain chelipeds, brought to the lab, and kept

in a tank supplied with a continuous flow of seawater. On Fridays, lobsters caught that week were examined and the following data recorded: sex, presence of eggs (berried), carapace length (CL), crusher claw position, missing claws and molt stage (Aiken 1973). Lobsters were then tagged with a serially numbered international orange sphyrion tag (Scarratt and Elson 1965; Scarratt 1970), and released at the site of capture. Recaptured tagged lobsters, severely injured or newly molted (soft) lobsters and those < 55 mm CL were not taken to the lab but returned to the water immediately after recording the above data.

Sampling at the effluent station was discontinued in 1978, due to the difficulty in hauling and keeping pots set properly in that area as strong currents and large boulders resulted in snagged trawls and lost pots. From 1975 through 1978 sampling was conducted from January to December and from 1979 through 1985 during the months of highest catch: May through October. Starting in 1979, surface and bottom water temperatures were recorded at each station.

To obtain more information on small lobsters, wire pots (2.5 cm^2) were added to the sampling design in August 1978. Half of the wood pots at each station were replaced with wire pots which were able to keep many of the small lobsters that were able to escape between the lath spaces of wood pots. Quantifying the abundance and population characteristics of these smaller individuals is important since they constitute the majority of prerecruits whose abundance largely determines the size of the legal catch. To further increase catch, all wood pots were replaced by wire pots in 1982 after completing a study (discussed under "Results and Discussion") to compare the performance of the two pot types.

In 1981, we began collecting additional data to determine the size at which females first become sexually mature. The maximum outside width of the second abdominal segment of all females was measured, to the nearest millimeter. Female size at sexual maturity was estimated by calculating the ratio of the abdominal width to the carapace length and plotting that ratio against the carapace length (Skud and Perkins 1969; Krouse 1973).

Beginning in 1982, pots were numbered individually to determine the variability in catch among pots. This information would provide more accurate values for catch-per-pot than an average catch-per-pot based on the 20 pots of each sampling location. In addition to recording the number of lobsters caught in each pot, we began counting the number of other organisms caught in pots to examine the influence of competing species on lobster catch. Catch per unit effort (CPUE) was adjusted by covariance analysis

for the effect of soaktime (number of days between pothauls) and the catch of competing species that significantly affected CPUE.

The size of the local lobster population was estimated during 1976-84 using the method of Jolly (1965) as modified by Seber (1965)(NUSCo 1984, 1985). This multiple census method uses tag and recapture data to estimate the size of the entire population at various points in time. Due to the low accuracy of the Jolly-Seber population estimates relative to the annual CPUE values, the use of Jolly-Seber estimates was discontinued in 1985 (see Appendix to NUSCo 1986). The annual CPUE (lobsters caught per pot) is a reliable index of relative population abundance, is interpreted more readily than a composite estimate, and has a reliable and well known estimate of variance.

Methods for the collection of lobsters on the intake traveling screens are described in the Fish Ecology section of this report under Methods and Materials-Impingement.

Initially, lobster larvae were enumerated from entrainment samples as part of the ichthyoplankton (IP) monitoring program (see Fish Ecology); however, the methods were not designed for sampling lobster larvae. Larger volumes of water must be sampled to adequately quantify lobster larvae density because of their patchy distribution in the water column. Beginning in 1984 a special lobster larvae entrainment study was initiated to provide better estimates of the number of lobster larvae entrained through the plant's cooling water systems. Lobster larvae sampling was conducted during the period of their occurrence (May through July) at Units 1 and 2 discharges. Samples were collected with a 1.0 x 6.0 m conical plankton net of 1.0 mm mesh deployed using a gantry system described previously (NUSCo 1978). Sample volumes were averaged from those calculated from the readings of four General Oceanic flowmeters. Four day and four night samples were collected weekly (1 day, 1 night on each of 4 days). Each sample was placed in a large 1.0 mm mesh sieve and kept in tanks supplied with a continuous flow of seawater. Samples were sorted shortly after collection in a white enamel pan and larvae were examined for movement and classified as either alive or dead. Lobster larvae were also classified by stage according to the criteria established by Herrick (1911) and stored in 70% ethyl alcohol.

Since some of the catch and population data collected prior to 1979 were incomplete, they do not provide long term continuity. Therefore, in some of the following Results and Discussion sections, only

the data which provided a comparable continuum were used to describe the lobster population under 2-unit operating conditions.

RESULTS AND DISCUSSION

Abundance and Catch Per Unit Effort

Annual catch statistics for lobsters caught in pots from 1976 to 1985 are presented in Table 2. A total of 51,717 lobsters was caught in 40,910 pothauls. Annual total catch per unit effort (CPUE) ranged from 0.56 to 2.10 lobsters per trap. The lower CPUE values for 1976-77 correspond to data collected with wood pots which allow small lobsters to escape between the 3-5 cm lath spacings.

Table 2. Catch statistics for lobsters caught in pots [a] from May through October (1976-85).

	Total Caught	Number Tagged	Number Recaptured	Pots Hauled	Total CPUE	Percent Recaptured
1976	1691	1503	188	3043	0.56	12.5
1977	1947	1773	168	3350	0.58	9.5
1978	3578	2768	521	4232	0.85	18.8
1979	5037	3732	722	4086	1.23	19.4
1980	4268	3634	522	4182	1.02	14.4
1981	5110	4246	704	4375	1.17	16.6
1982	9109	7575	1278	4340	2.10	16.9
1983	6376	5160	936	4285	1.49	18.1
1984	7587	5992	1431	4550	1.67	23.9
1985	7014	5609	1235	4467	1.57	22.2
1976-1985	51717	41992	7705	40910	1.26	18.4

[a] 60 wood pots used from 1976-77; 30 wood and 30 wire pots used from 1978-81; 60 wire pots used from 1982-85.

Wood and wire pots used during a 3 1/2 yr gear-comparison study provided the basis for using all wire pots in our lobster studies beginning in 1982. Monthly total and legal CPUE for wood and wire pots are presented in Table 3. The CPUE data are proportions (no. lobsters caught/no. pots hauled) with nonhomogeneous variances and nonnormal distributions. Therefore, the nonparametric Wilcoxon 2-sample test was used to test for equal catchability between pot types. The CPUE of legal-sized lobsters (greater than or equal to 81 mm CL) was similar for wood and wire pots throughout the gear comparison study; however, except for 1981, the total CPUE was significantly higher for wire pots. This, of course, was due to greater catches of sub-legal lobsters in the wire pots.

Table 3. Monthly catch per unit effort for wood and wire pots during gear comparibility study.

Year	Month	Total CPUE		Legal CPUE	
		Wood	Wire	Wood	Wire
1978	Aug	0.55	2.15	0.08	0.18
	Sept	0.96	1.77	0.14	0.19
	Oct	0.54	1.32	0.08	0.15
	Nov	0.98	1.62	0.16	0.12
1979	May	0.86	1.32	0.06	0.06
	June	1.03	1.83	0.18	0.15
	July	1.24	1.95	0.23	0.25
	Aug	0.95	1.64	0.15	0.12
	Sept	0.69	1.51	0.10	0.09
	Oct	0.55	1.12	0.09	0.09
1980	May	0.79	1.80	0.15	0.15
	June	0.65	1.70	0.14	0.12
	July	0.69	1.77	0.18	0.17
	Aug	0.56	1.31	0.13	0.12
	Sept	0.69	0.84	0.12	0.08
	Oct	0.78	0.74	0.06	0.03
1981	May	1.18	1.34	0.08	0.08
	June	1.53	1.16	0.15	0.08
	July	1.57	1.24	0.21	0.13
	Aug	1.22	1.00	0.13	0.11
	Sept	1.22	0.73	0.11	0.10
	Oct	1.12	0.66	0.13	0.09

Summary of Wilcoxon 2-sample tests for equal catch regardless of pot type.

	z-statistics[a] for total CPUE	z-statistics[a] for legal CPUE
1978	$z = -2.16$ ($p < 0.05$)	$z = -1.31$ ($p = 0.19$)
1979	$z = -2.64$ ($p < 0.01$)	$z = 0.24$ ($p = 0.81$)
1980	$z = -2.49$ ($p < 0.05$)	$z = 0.65$ ($p = 0.52$)
1981	$z = 1.36$ ($p = 0.17$)	$z = 1.63$ ($p = 0.10$)

[a] The z-statistic corresponds to the standard normal distribution.

A special study was initiated during 1982 to investigate the lower catch of wire pots used in 1981 and to examine trap efficiency in relation to the construction of parlor entry funnels in wood and wire pots. The wire pots used in the 1981 study (5 from each station) were fished with 15 newly constructed wire pots during the 1982 sampling period. The old wire pots (used in 1981) were fished without any modification during the first 6 weeks of the study. These old pots were then removed for 1 week to install new parlor and entry funnels. They were then deployed as before and fished for the remainder of the study.

In Figure 2, the plot of the weekly catch in the old and new wire pots shows a great disparity between the catch efficiency in the two pot types before potheads were changed (prior to week 6). However, after the parlor entry funnels were changed in these 2 yr old pots they fished as well as newly constructed wire pots. These results support the observations of Spurr (1972) and Thomas (1959) further demonstrating the sensitivity of trap efficiency to parlor head design and placement. Spurr found that the principal factor affecting pot efficiency is parlor funnel design and Thomas found that high-rigged entry funnels deterred escape.

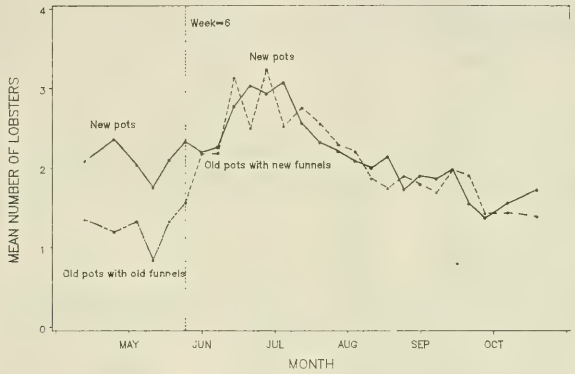

Figure 2. Weekly mean number of lobsters caught with old pots (old funnels vs. new funnels) and new pots at all stations during 1982.

Other factors we examined that contribute to the efficiency of lobster traps are the number of days between pothauls (soaktime) and the influence of competing species caught in traps. The effectiveness of bait attracting lobsters into a trap is reduced when traps are set out for several days without rebaiting. The bait deteriorates more rapidly in the warmer waters of summertime than in spring or fall. Accordingly, lobstermen adjust the timing of pothauls to ensure that pots always have an ample supply of bait. Competing species caught in traps also feed on the bait. Lobstermen alter their pothaul schedules over the year to ensure that their traps do not become overcrowded. Generally the influence of soaktime and

competing species are more important during the summer months when water temperatures and crustacean activity (feeding, molting, reproduction, etc.) are at a maximum.

The incidental catch of spider crabs (*Libinia* spp.) during 1984 and hermit crabs (*Pagurus* spp.) and cunner (*Tautogolabrus adspersus*) during 1985, had the greatest influence on lobster catch of all competing species caught in traps (Table 4).

Table 4. Total numbers of lobsters and incidental species caught at each station from May to October, during 1984 and 1985.

	Jordan Cove		Intake		Twotree	
	1984	1985	1984	1985	1984	1985
Lobster	2657	2257	2238	1383	2692	3374
Rock crab	71	16	208	87	112	42
Jonah crab	12	2	26	1	36	29
Spider crab[a]	437	163	2729	1721	71	66
Hermit crab[b]	28	35	323	252	77	209
Blue crab	7	8	32	13	1	0
Winter flounder	34	32	8	5	3	4
Summer flounder	24	9	27	15	9	0
Skates	2	4	6	4	7	9
Oyster toadfish	38	27	37	40	1	0
Scup	3	34	23	29	1	27
Cunner[b]	33	22	28	10	80	175
Tautog	1	16	27	118	11	116
Sea raven	7	9	11	6	2	4
Whelks	4	1	21	3	41	74

[a] Species with significant ($p < 0.05$) effect on CPUE in 1984.
[b] Species with significant ($p < 0.05$) effect on CPUE in 1985.

Since the effects of soaktime, the incidental catch of spider crabs in 1984 and hermit crabs and cunner in 1985 significantly biased the values for lobster CPUE, we adjusted our mean monthly CPUE through covariance analysis. Soaktime, the number of spider crabs caught in 1984 and the number of hermit crabs and cunner caught in 1985 were used as covariates. Monthly catches (CPUE, and CPUE adjusted for the covariates) are presented for each station for 1984 and 1985 in Table 5.

Table 5. Catch statistics for lobsters caught at each station from May to October during 1984-85.

1984

Month	Number of pots hauled	Total number caught	Total CPUE Actual	Total CPUE Adjusted[a]	Total legals caught	Legal CPUE
			Jordan Cove			
MAY	240	496	2.07	2.07	28	0.12
JUN	260	503	1.94	1.95	29	0.11
JUL	240	530	2.21	2.16	42	0.18
AUG	280	493	1.76	1.78	33	0.12
SEP	219	303	1.38	1.33	24	0.11
OCT	255	332	1.30	1.23	15	0.06
			Intake			
MAY	240	309	1.29	1.50	20	0.08
JUN	260	486	1.87	1.90	37	0.14
JUL	239	515	2.16	2.17	36	0.15
AUG	280	397	1.42	1.44	22	0.08
SEP	220	249	1.13	1.09	20	0.09
OCT	260	282	1.09	1.05	12	0.05
			Twotree			
MAY	239	436	1.82	1.80	84	0.35
JUN	260	443	1.70	1.72	59	0.23
JUL	240	438	1.83	1.77	67	0.28
AUG	279	579	2.08	2.08	70	0.25
SEP	220	382	1.74	1.68	56	0.26
OCT	259	414	1.60	1.55	56	0.22

[a] CPUE values adjusted for the effects of soaktime, and the catch of spider crabs

1985

Month	Number of pots hauled	Total number caught	Total CPUE Actual	Total CPUE Adjusted[b]	Total legals caught	Legal CPUE
			Jordan Cove			
MAY	258	484	1.88	1.83	34	0.13
JUN	238	430	1.81	1.78	36	0.15
JUL	280	566	2.02	2.01	33	0.12
AUG	258	310	1.20	1.19	16	0.06
SEP	200	177	0.89	0.83	8	0.04
OCT	260	290	1.12	1.10	12	0.05
			Intake			
MAY	260	317	1.22	1.24	12	0.05
JUN	218	284	1.30	1.19	35	0.16
JUL	279	337	1.21	1.20	22	0.08
AUG	257	180	0.70	0.73	5	0.02
SEP	200	89	0.45	0.43	2	0.01
OCT	260	176	0.68	0.70	5	0.02
			Twotree			
MAY	260	600	2.31	2.28	40	0.15
JUN	239	638	2.67	2.66	47	0.20
JUL	280	927	3.31	3.30	94	0.34
AUG	260	514	1.98	2.06	32	0.12
SEP	200	302	1.51	1.52	21	0.11
OCT	260	393	1.51	1.52	20	0.08

[b] CPUE values adjusted for the effects of soaktime, and the catches of hermit crabs and cunner.

Annual catch statistics for lobsters caught in wire pots at each station from 1978 to 1985 are presented in Table 6. The total CPUE was greatest at Twotree (range 1.15-2.52) followed by Intake (0.94-2.07) and Jordan Cove (0.88-1.91). CPUE of legal-sized lobsters was also greatest at Twotree (0.11-0.28) followed by Intake (0.05-0.22) and Jordan Cove (0.07-0.15). The total CPUE from Intake during 1985 was the lowest value reported for that station since wire pots were first used in 1978. Only 20% of the 1985 catch was taken at Intake, whereas Intake catch comprised 26-40% of the total catch from 1978 to 1984. Dredging activities in the vicinity of the intake structures during June 1985 were responsible for the lower Intake catch. Because dredging removes existing habitat (shelters), lobsters are displaced temporarily until the dredged area stabilizes. After the dredged area has stabilized lobsters will probably return to the area and catch rates at that station should increase in 1986. Effects of dredging on the benthic infaunal community at Intake have also been documented in the Benthic Infauna section of this report.

Table 6. Catch statistics for lobsters caught in wire pots at each station from 1978 through 1985.

	Number of pots hauled	Total number caught	Total CPUE	Total legals caught	Legal CPUE
			Jordan Cove		
1978	349	634	1.82	34	0.10
1979	701	1337	1.91	97	0.14
1980	722	966	1.34	63	0.09
1981	724	640	0.88	51	0.07
1982	1473	2816	1.91	152	0.10
1983	1449	2368	1.63	218	0.15
1984	1494	2657	1.78	171	0.12
1985	1494	2257	1.51	139	0.09
			Intake		
1978	348	720	2.07	77	0.22
1979	709	1184	1.67	98	0.14
1980	721	903	1.25	60	0.08
1981	730	749	1.03	39	0.05
1982	1449	2740	1.89	153	0.11
1983	1439	1646	1.14	126	0.09
1984	1499	2238	1.49	147	0.10
1985	1474	1383	0.94	81	0.06
			Twotree		
1978	329	470	1.43	67	0.20
1979	641	738	1.15	72	0.11
1980	673	987	1.47	109	0.16
1981	733	847	1.16	127	0.17
1982	1418	3567	2.52	403	0.28
1983	1456	2350	1.61	308	0.21
1984	1497	2692	1.80	392	0.26
1985	1499	3374	2.25	254	0.17

Annual CPUE's and approximate 95% C.I. for wire pot data are presented in Table 7. These CPUE's were computed as the geometric mean of the ratios of (no. lobsters) to (no. wire pots), because these ratios are nonadditive and have an asymmetric distribution about the arithmetic mean. Therefore, the geometric mean is the best statistic for constructing C.I.'s which should be asymmetric. Based on the inspection of the C.I.'s in Table 7, the 1980 and 1981 CPUE's were the lowest annual values of all years from 1978 to 1985. Conversely, the 1982 CPUE was the highest annual value when all years were compared and was due to a strong prerecruit class in 1982 (also apparent in the 1982 frequency distribution - see next section) which sustained record landings in 1983 and 1984 when these lobsters molted to legal size.

Table 7. Geometric mean CPUE and approximated upper and lower 95% confidence intervals for lobsters caught in wire pots at all stations from 1978 through 1985.

Year	N	Lower Bound	CPUE	Upper Bound
1978	104	1.454	1.600	1.761
1979	208	1.302	1.404	1.513
1980	218	0.997	1.103	1.221
1981	220	0.839	0.904	0.974
1982	220	1.925	2.006	2.089
1983	218	1.250	1.331	1.417
1984	225	1.540	1.607	1.677
1985	221	1.252	1.352	1.460

Population Characteristics

Size Frequencies

Annual size frequency distributions for male and female lobsters caught in wire pots from 1979 to 1985 are shown in Figure 3 and for each station in Figure 4. Population statistics for lobsters caught from 1976 to 1985 are summarized in Table 8. Annual mean CL's have been consistent since lobsters have been collected using pots. The mean CL of lobsters caught in wood pots was greater (range 73.3-76.6) than the mean CL of lobsters caught in wire pots (range 70.8-71.8).

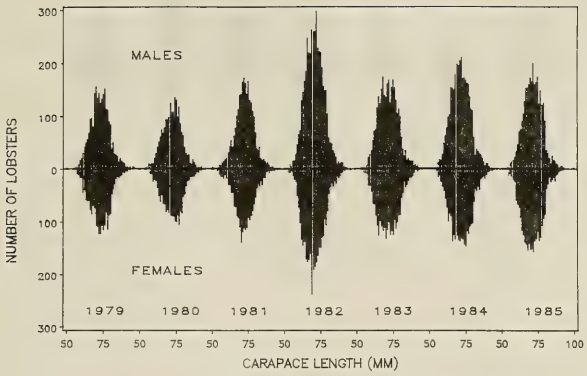

Figure 3. Size frequency distributions for male and female lobsters caught at all stations from 1979 through 1985.

An important objective of this study was to gather information on as large a segment of the local lobster population as possible. Through the use of wire pots (2.5 cm^2 mesh) we anticipated increased catch of smaller sized lobsters capable of escaping through the 3-5 cm gap between the laths of the commercial wood pots. A Kolmogorov-Smirnov test on the size frequency distributions of lobsters caught in the two pot types indicated that wire pots caught significantly ($p < 0.05$) more of the size class smaller than 75 mm CL than did wood pots. These results are similar to those of Krouse (1973) who found that the CL of the catch from wire pots averaged between 67.9 and 70.5 mm. He considered the modal size of his catch (70 mm CL) to be the size at which lobsters are less apt to escape the traps; using the same reasoning, lobsters in our study were vulnerable to the wire pots at 70 mm CL and to the wood pots at about 76 mm CL. Knowledge of lobsters in the 70-76 mm size class is important, since these individuals constitute a large proportion of the prerecruits (i.e., those individuals within one molt of legal size). Since the lobster population of the Millstone Point region is subjected to a high exploitation rate (Keser et al. 1983), the size of the legal catch is largely determined by the abundance of the prerecruit size class the year before. The sensitivity of our sampling effort in defining year class strength was apparent in 1982. As stated in the previous section, during 1982, we observed a very strong prerecruit class (Fig. 3); when these individuals molted the following year, record landings were realized throughout LIS in 1983 and 1984 (CT DEP Marine Fisheries Statistics).

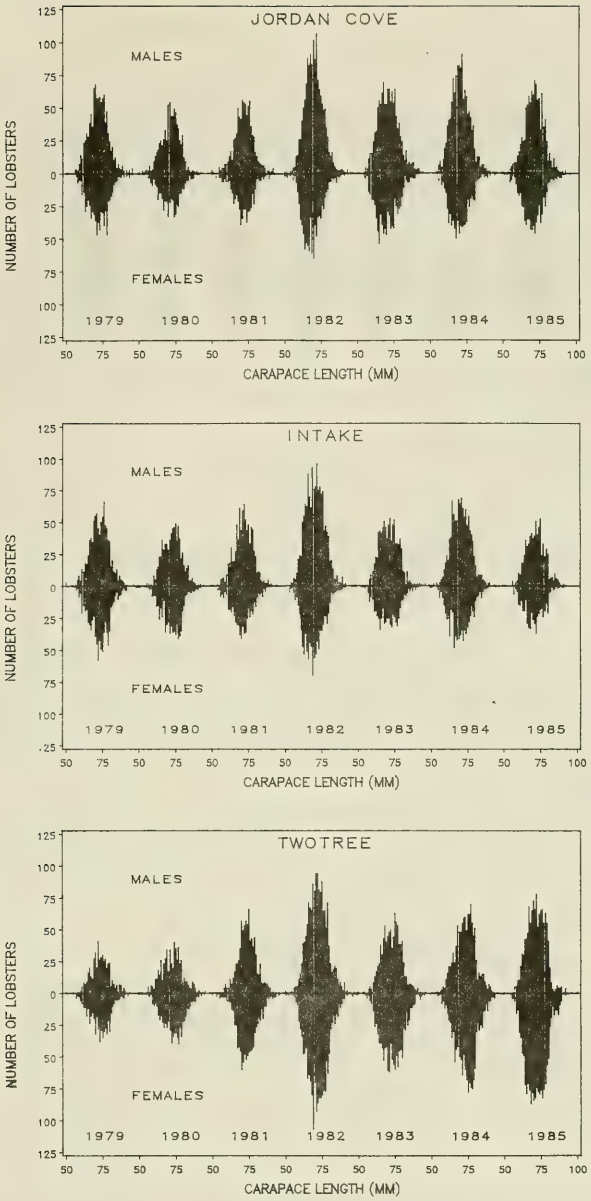

Figure 4. Size frequency distributions for male and female lobsters caught at each station from 1979 through 1985.

Table 8. Population statistics for lobsters caught in wood and wire pots from 1976 to 1985.

	WOOD			WIRE		
	N[a]	Mean CL ± 1 S.D.	Percent Legals	N[a]	Mean CL ± 1 S.D.	Percent Legals
1976	596	73.6 ± 7.0	13.6			
1977	549	76.6 ± 6.6	15.7			
1978	777	75.1 ± 5.6	14.2	1910	71.5 ± 6.5	8.5
1979	1510	75.7 ± 5.6	15.4	2846	71.2 ± 7.0	8.2
1980	1213	75.9 ± 5.8	18.1	2529	70.9 ± 6.6	7.2
1981	2423	73.3 ± 6.0	10.4	1983	71.0 ± 7.5	9.6
1982				7839	70.8 ± 6.9	6.3
1983				5435	71.7 ± 7.3	10.1
1984				6156	71.8 ± 7.1	9.6
1985				5723	71.3 ± 6.7	6.6

[a] Recaptures not included

Sex Ratios

Since 1975, the overall sex ratio of males to females was close to 1:1 (Table 9). However, when three stations were compared, Twotree had consistently higher proportions of females, whereas Intake and Jordan Cove had slightly more males. Sex ratios close to 1:1 were also reported by other researchers working in waters close to shore (Herrick 1911; Templeman 1936; Ennis 1971, 1974; Stewart 1972; Krouse 1973; Thomas 1973; Cooper et al. 1975; Briggs and Mushacke 1980). Smith (1977) found male to female ratios of the commercial catch ranging from 1:1.06 to 1:1.81 in four different areas of LIS which agrees with the ratios of our Twotree station which is 1.5 km offshore. Ennis (1980) indicated that sex ratios are dependent on the size composition of the catch which in turn is dependent on the method and depth of sampling. Ratios close to 1:1 occur up to the size at which females are sexually mature, after which females tend to predominate in the catch due to variation in trapping behavior related to molting and reproduction, legal restrictions of taking egg-bearing females, and the fact that mature females molt less frequently than males (Skud and Perkins 1969; Cooper et al. 1975; Ennis 1980).

Table 9. Male to female sex ratios of lobsters caught at each station from 1975 to 1985.

	Jordan Cove	Intake	Twotree
1975	1.0 : 0.71	1.0 : 0.86	1.0 : 1.55
1976	1.0 : 0.76	1.0 : 0.97	1.0 : 1.83
1977	1.0 : 0.75	1.0 : 0.89	1.0 : 1.27
1978	1.0 : 0.75	1.0 : 0.95	1.0 : 1.13
1979	1.0 : 0.70	1.0 : 0.87	1.0 : 1.13
1980	1.0 : 0.68	1.0 : 0.87	1.0 : 1.18
1981	1.0 : 0.65	1.0 : 0.68	1.0 : 1.13
1982	1.0 : 0.62	1.0 : 0.66	1.0 : 1.13
1983	1.0 : 0.70	1.0 : 0.65	1.0 : 1.28
1984	1.0 : 0.60	1.0 : 0.68	1.0 : 1.27
1985	1.0 : 0.70	1.0 : 0.66	1.0 : 1.38

Reproductive Activities

The presence of external eggs indicates that females are mature and the size at onset of maturity can be determined from the size distribution of berried females. Another method of determining female size at sexual maturity was described initially by Templeman (1935) who observed that the relative width of the second abdominal segment of females increased with the approach of sexual maturity. In our study we measured the second abdominal segment widths of all females, calculated the ratio of the abdominal width to the carapace length, and plotted that ratio against the carapace length (Skud and Perkins 1969; Krouse 1973).

The morphometric relationship between carapace length and abdominal width for data collected from 1981 to 1985 is described in Figure 5.

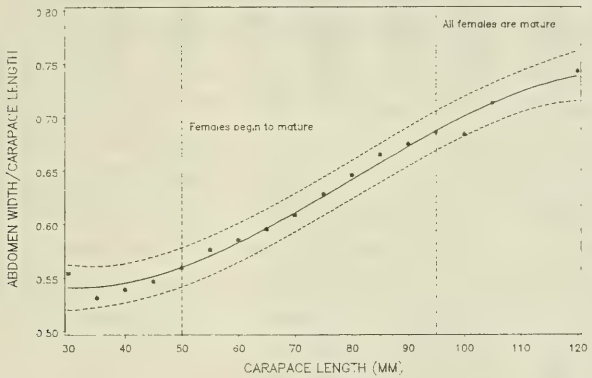

Figure 5. Morphometric relationship between the ratio abdominal width/carapace length and the carapace length for data collected from 1981 to 1985 for female lobsters.
(*) mean value for each 5 mm size; (——) $y = a + bx + cx^2 + dx^3$; (- - -) upper and lower 95% C.I.

The carapace length and abdomen width increase in size proportionately up to the size at which females begin to mature (about 50 mm CL for our area) after which the abdomen increases in width faster than the carapace increases in length. When all females are mature (about 95 mm CL) the relationship between the carapace length and the abdominal width is again proportional. In western LIS, females begin to

mature at about 60 mm CL and most are mature at about 80 mm CL (Briggs and Mushacke 1979). In contrast, outside of LIS, females begin to mature at sizes substantially larger than in our area. In northern areas low water temperatures retard reproductive maturation, whereas warmer summer water temperatures of LIS favor early maturation of females (Smith 1977; Aiken and Waddy 1980).

Berried females have comprised between 3.1 and 6.7% of all females caught from 1975 to 1985. Females predominate at Twotree and greater proportions of berried females were caught there (5.1-10.1%) when compared to Jordan Cove (1.0-4.5%) and Intake (1.1-4.3%) (Table 10).

Table 10. Percentage of berried females at each station and mean carapace lengths from 1975 to 1985.

	Jordan Cove	Intake	Twotree	N[a]	Length Range (mm)	Length Mean±1 SD
1975	4.5	3.5	9.7	7	73 - 84	79.1 ± 3.7
1976	2.0	3.3	11.2	16	70 - 102	82.9 ± 7.7
1977	1.4	3.5	6.2	35	68 - 92	79.7 ± 6.4
1978	1.4	2.5	5.1	58	74 - 88	80.1 ± 4.0
1979	1.9	2.7	6.2	67	64 - 93	80.6 ± 5.4
1980	3.4	1.8	5.4	71	72 - 93	79.2 ± 5.1
1981	1.8	2.7	6.9	82	70 - 97	81.2 ± 6.1
1982	1.0	1.1	6.2	108	64 - 99	80.0 ± 5.8
1983	2.5	3.6	8.3	123	66 - 103	80.5 ± 5.9
1984	4.3	3.3	10.1	173	62 - 95	79.1 ± 5.8
1985	3.5	4.3	8.0	171	63 - 94	77.0 ± 5.4

[a] Recaptures not included

The size distribution (range, mean CL) of berried females collected have provided further evidence for the small size at which females become mature in LIS (Table 10). The smallest berried females collected (62-64 mm CL) were between 54-56 mm CL when oviposition first occurred assuming 14% growth per molt. This confirms our carapace length/abdominal width relationship and suggests that 50 mm CL is the size at which females begin to sexually mature in LIS.

The fact that females mature at a small size in LIS and that over half of the berried females are sublegal size is important because these individuals are able to spawn before growing to marketable size (Fig. 6).

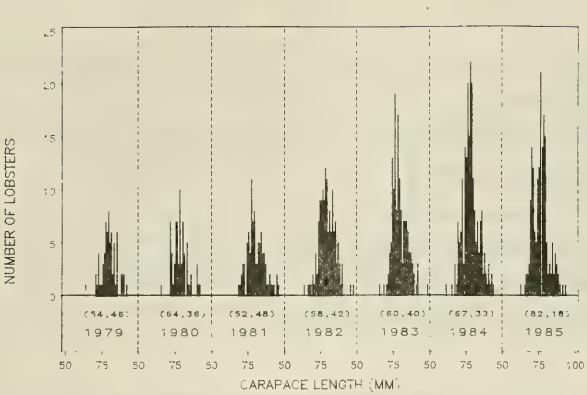

Figure 6. Size frequency distribution for berried females caught from 1979 to 1985. Numbers in parentheses represent the percentages of sublegal and legal berried females (%sublegal, %legal).

In comparison, females in northern and offshore populations (Maine, Canada) begin to mature at sizes close to the legal size and only a small percentage is able to spawn prior to reaching marketable size (Aiken and Waddy 1980).

To provide more information about the reproductive cycle of lobsters in our area, we recorded both the fullness, and the developmental stage of egg masses carried by berried females during 1984-85 (Table 11). Based on embryo development, we concluded that berried females caught in May and June carried eggs that were ready to hatch. The small number of berried females caught in July indicated the completion of the biennial spawning cycle. Females that were fertilized in the previous year began extruding eggs in August and the number of berried females caught carrying newly extruded eggs peaked in September and October. About 89% of the berried females examined in 1984 for egg mass fullness had 1/2 or more the normal complement of eggs and, in 1985, 86% had 1/2 or more the normal complement. Only 3.7% of the berried females in 1984 and, in 1985, 7.7% had less than 1/4 the normal complement of eggs. This may be compared to 10-14% of the berried females in western LIS carrying abnormally low numbers of eggs in 1976 (Smith 1977). Smith's concern was that such an additional source of natural mortality (i.e., abnormally low fecundity) in western LIS, an area where 30% of the females are berried, could affect the entire Connecticut fishery.

Table 11. The number of berried females examined for egg mass fullness and egg development from May to October during 1984-85.

Month	Number of Berried Females Examined	Number with $< \frac{1}{4}$ Complement	Number with $\frac{1}{4}$ Complement	Number with $\frac{1}{2}$ Complement	Number with $\frac{3}{4}$ Complement	Number with Full Complement	Developmental Stage
			1984				
MAY	28	0	1	4	11	12	l.t. Green with optical disks
JUN	16	4	1	2	1	8	"
JUL	4	0	0	1	1	2	"
AUG	18	0	0	0	4	14	Black-dark green
SEP	48	1	5	16	10	16	"
OCT	50	1	5	7	16	21	"
TOTAL	164	6	12	30	43	73	
			1985				
MAY	34	3	4	8	3	16	l.t. Green with optical disks
JUN	19	6	1	0	5	7	"
JUL	7	2	0	0	4	1	"
AUG	17	0	1	1	3	12	Black-Dark Green
SEP	56	2	3	4	14	33	"
OCT	89	11	6	8	23	48	"
TOTAL	222	17	15	21	52	117	

Molting and Growth

Lobster growth was determined from carapace length measurements for those lobsters that molted between tagging and recapture. The number of molting lobsters observed in the weekly catch varied from year to year and over the sampling period (Fig. 7). In general, molting peaked in June although in several years a fall molting peak was also observed. Frequency of molting and size increase per molt are reported to be affected by temperature, light, nutrition, social behavior, injury/regeneration, habitat, season of year and reproductive development (Aiken and Waddy 1980). The fact that secondary molts were observed in our study area is not unusual; two molting peaks were observed by Lund et al. (1973) for LIS and by Russell et al. (1978) for Narragansett Bay.

Several researchers have shown that growth increment per molt in crustaceans is best described by linear regression (Wilder 1953; Kurata 1962; Mauchline 1976). Carapace lengths at recapture (post-molt size) were regressed on carapace lengths at tagging (pre-molt size) for data collected from 1978 to 1985. Regression plots, equations and growth parameters for all lobsters ($n = 733$) and males ($n = 296$) and females ($n = 437$) are presented in Figure 8.

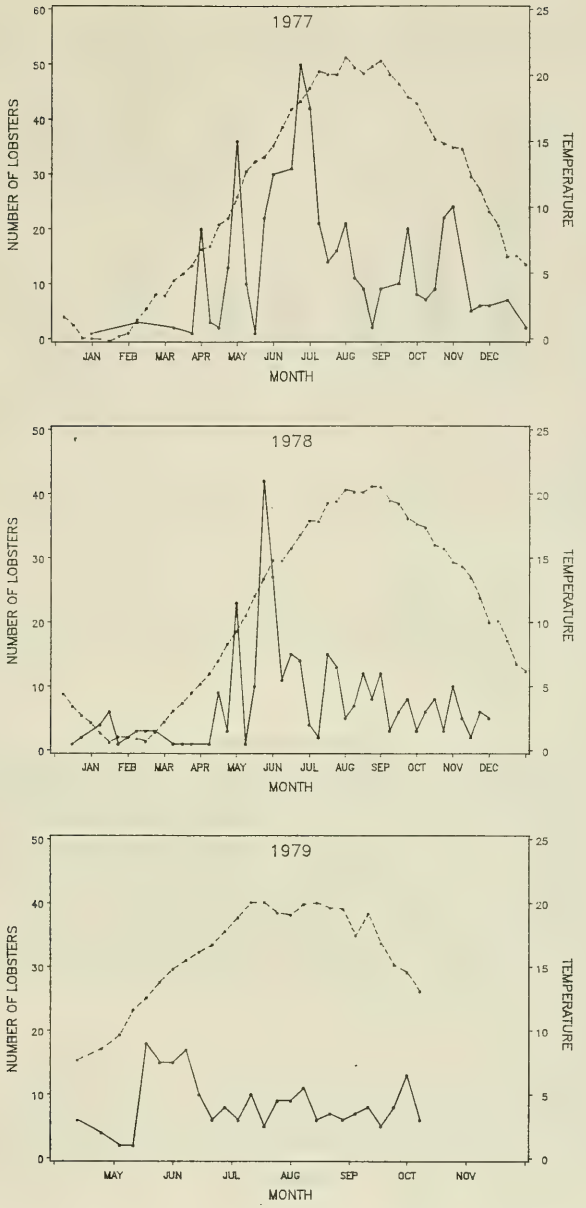

Figure 7. The number of molting lobsters (——) and bottom water temperature (- - -) from 1977 to 1985.

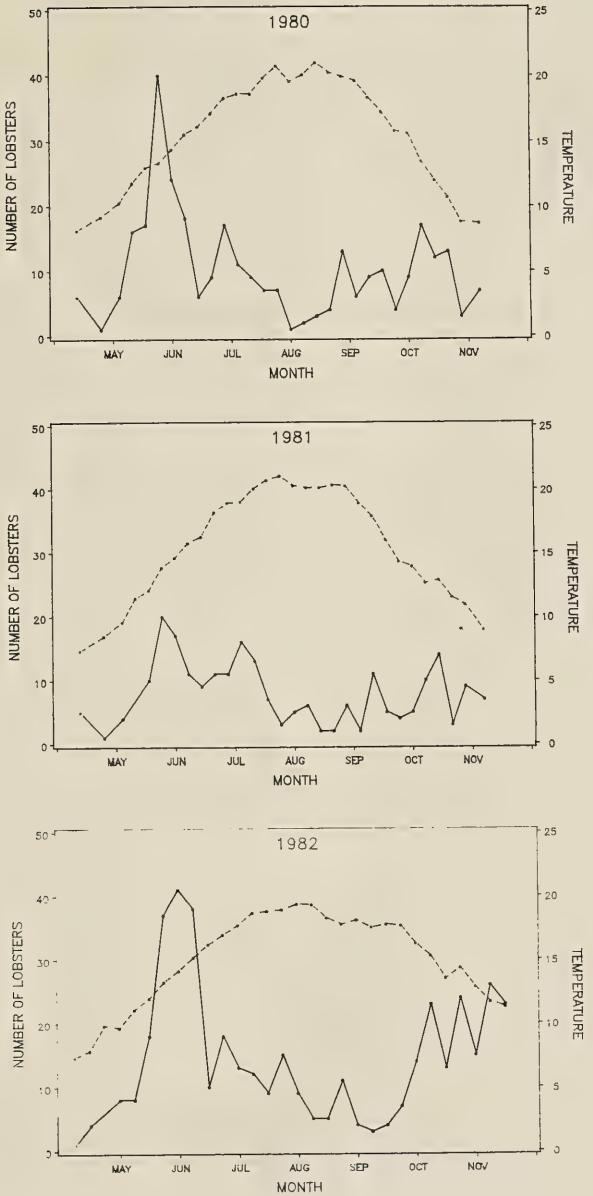
1980
1981
1982
NUMBER OF LOBSTERS
TEMPERATURE
MONTH
MAY
JUN
JUL
AUG
SEP
OCT
NOV
0
10
20
30
40
50
0
5
10
15
20
25

Fig. 7. cont.

1983
1984
1985
NUMBER OF LOBSTERS
TEMPERATURE
MONTH
MAY
JUN
JUL
AUG
SEP
OCT
NOV
0
5
10
15
20
25
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40
50

Fig. 7. cont.

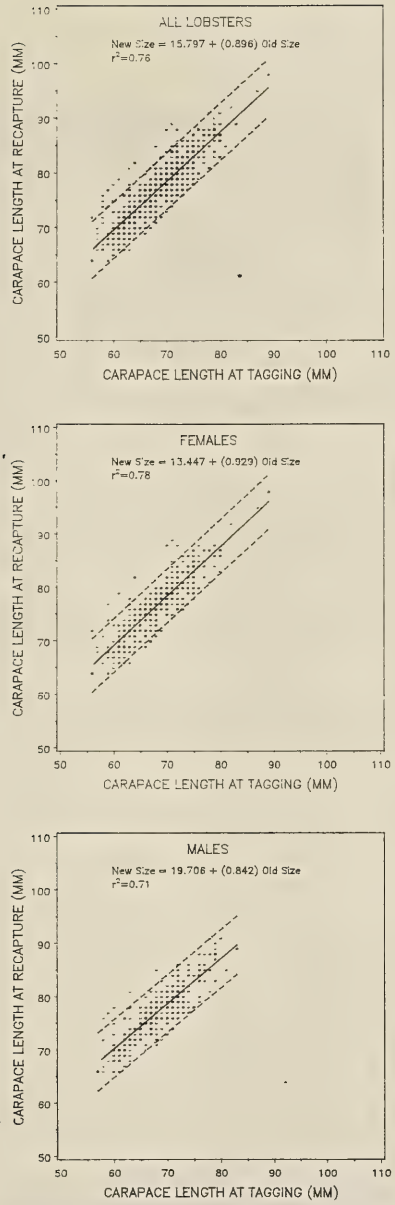
ALL LOBSTERS
New Size = 15.797 + (0.896) Old Size
r²=0.76
CARAPACE LENGTH AT RECAPTURE (MM)
CARAPACE LENGTH AT TAGGING (MM)
FEMALES
New Size = 13.447 + (0.929) Old Size
r²=0.78
CARAPACE LENGTH AT RECAPTURE (MM)
CARAPACE LENGTH AT TAGGING (MM)
MALES
New Size = 19.706 + (0.842) Old Size
r²=0.71
CARAPACE LENGTH AT RECAPTURE (MM)
CARAPACE LENGTH AT TAGGING (MM)

Figure 8. Linear regressions for carapace lengths at tagging and recapture for all lobsters, males and females for data collected from 1978 through 1985. (——) $y = a + bx$; (- - -) upper and lower 95% C.I.

Growth of males and females was significantly different based on t-tests of y-intercepts and slopes ($p > 0.05$). Smaller-sized males (< 60 mm CL) had a greater incremental size increase when compared to females of the same size (Fig. 9).

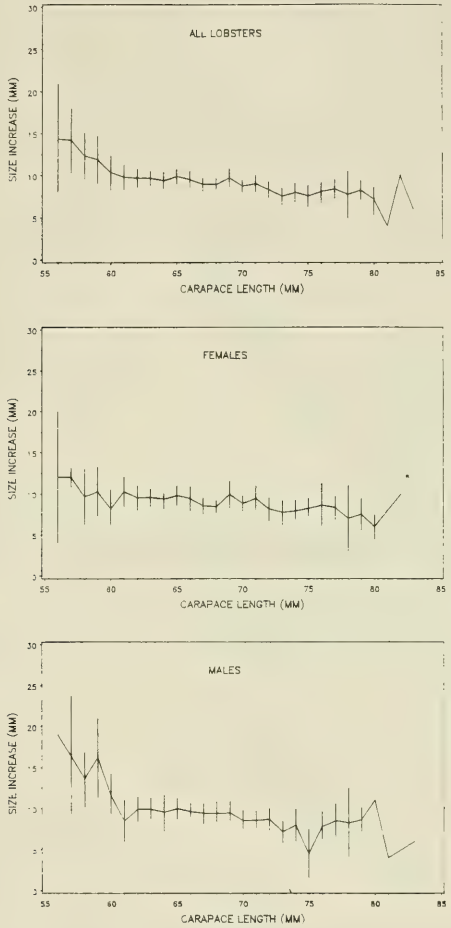

Figure 9. Mean incremental size increase for all lobsters, males and females from tag and recapture data collected from 1978 through 1985 (vertical bars = 2 standard deviations).

Lower incremental size increases for females smaller than 60 mm (CL) are expected since females begin to sexually mature at about 50 mm. Energy that would otherwise have been used in carapace growth was diverted to widening of the abdominal segments and development of the ovaries. The average growth per molt for males (14.1%) and females (13.7%) was similar to results reported by Briggs and Mushacke (1984) for western LIS lobsters (males 14.5%; females 12.5%). Cooper and Uzmann (1980) reported higher growth increments for the offshore lobster population, 18.7% for males and 16.7% for females. They attributed the lower growth of the inshore population to lobster inactivity during the colder months of the year.

Claw Loss

The percentage of lobsters missing one or both claws (culls) ranged from 9.0 to 17.4% for lobsters caught in wood pots and 10.6 to 15.5% for lobsters caught in wire pots (Table 12).

Table 12. Claw loss for lobsters caught in wood and wire pots from 1975 to 1985.

	Percent Cull		Percent Missing One Claw		Percent Missing Two Claws	
Year	Wood	Wire	Wood	Wire	Wood	Wire
1975	9.0	-	7.8	-	1.3	-
1976	15.4	-	13.5	-	2.0	-
1977	11.6	-	10.4	-	1.2	-
1978	15.9	14.9	14.1	14.0	1.9	0.9
1979	17.4	15.5	15.0	14.4	2.5	1.2
1980	17.0	13.6	14.8	12.2	2.2	1.5
1981	14.3	12.0	13.0	11.1	1.3	1.0
1982	-	11.1	-	10.4	-	0.7
1983	-	12.4	-	11.6	-	0.8
1984	-	10.6	-	9.8	-	0.7
1985	-	11.1	-	10.4	-	0.7

These percentages are typical for LIS; Smith (1977) reported 26.4% claw loss in LIS east of the Connecticut River and Briggs and Mushacke (1979) reported claw loss varying between 7.4 and 22.8% in western LIS. In general, the proportion of culls at each station was similar, and lobsters with missing or damaged claws were observed more frequently after the spring and fall molts. Pecci et al. (1978) reported that trap-related injuries resulting in claw loss are often associated with water temperature, fishing pressure (i.e., handling by lobstermen), trap soaktime, and physical condition of the lobster (i.e., its nearness to molting).

Tagging Program

From 1975 to 1985, 57,359 lobsters were caught; 47,259 were tagged and released, and 8,053 (17%) subsequently were recaptured in our sampling program (Table 13).

Table 13. Summary of tag and recapture studies from 1975 through 1985.

			NUSCo			Commercial		
Year	No. tagged	Mean CL (mm)	Returns No.	Pct.	Mean CL (mm)	Returns No.	Pct.	Mean CL (mm)
1975	2285	67.3	123	5.4	[a]	116	5.1	80.2
1976	2963	75.5	351	11.9	[a]	515	17.4	79.6
1977	2870	74.5	240	8.4	[a]	311	10.8	79.3
1978	3193	73.6	508	15.9	75.5	884	27.7	81.1
1979	3732	72.8	722	19.4	75.1	1776	47.6	77.5
1980	3634	75.5	522	14.4	75.7	1363	37.5	76.4
1981	4246	72.4	707	16.7	74.8	1484	35.0	76.3
1982	7575	70.9	1278	16.9	73.2	2518	33.2	75.5
1983	5160	71.8	936	18.1	73.6	2266	43.9	76.9
1984	5992	71.9	1431	23.9	73.0	1262	21.1	78.7
1985	5609	71.3	1235	22.0	73.1	899	16.0	78.3

[a] Recaptures not measured.

Commercial lobstermen caught an additional 13,394 (28.3%) of our tagged lobsters over the same period. In recent years (1984-85), the percentage of tagged lobsters caught by commercial lobstermen was lower (18.6%) than rates of recapture from 1975 to 1983 (30.6%). Conversely, our rates of recapture during 1984-85 were higher (23.0%) than rates during 1975-83 (15.1%). This disparity in rates of recapture was due to the implementation of the escape vent regulation in April 1984 (Landers and Blake 1985). The escape vent (1 3/4 in x 6 in) allows the escape of sublegal sized lobsters and, since the majority of our tagged lobsters are sublegals, fewer were retained in the commercial traps with the required vents. In comparison, our traps do not contain escape vents and retain greater numbers of tagged sublegals.

Additional evidence for the effectiveness of escape vents was apparent in the mean size of tagged lobsters caught in commercial gear before (1975-83) and after implementing the escape vent regulation in

1984-85. The mean CL of tagged lobsters caught in commercial gear prior to the escape vent regulation was 76.5 mm and was significantly smaller (t-test $p < 0.01$) than the mean CL of 78.6 mm for tagged lobsters caught with the regulation in force. Krouse and Thomas (1975) calculated selectivity curves for lobsters caught in traps with various vent sizes (1 1/2, 1 5/8, 1 3/4 in). Traps fitted with a 1 3/4 in escape vent had 50% retention of lobsters ranging in size from 75.4-78.8 mm CL. The mean CL of lobsters caught in our wire traps since 1978 ranged between 70.8 and 71.8 mm and was lower than the 50% retention sizes reported by Krouse and Thomas (1975). Thus, most of our tagged lobsters are able to escape from vented commercial traps thereby decreasing the probability of their recapture in commercial traps and increasing the proportions captured by unvented pots.

Movement

Because lobsters were tagged and released at the station where captured, any migrations from the capture area could be detected at recapture. Recapture data from our sampling efforts and those of commercial lobstermen were used to assess movement patterns.

About 95% of the lobsters were recaptured at the release station with movements between stations being minimal. Of the exchanges that did occur, most were between Jordan Cove and Intake. During 1976-77, when sampling was conducted at the Effluent station, only 58% of the lobsters released there were subsequently recaptured there. Thirty percent moved to Jordan Cove, 9% moved to the Intake station and 2.5% moved to Twotree. This suggests that lobsters may visit the effluent area to feed since it is a productive area for mussels and other benthic invertebrates.

Tagging studies conducted in coastal waters of eastern North America indicate localized lobster movement (Templeman 1935, 1940; Wilder and Murray 1958; Wilder 1963; Cooper 1970; Stewart 1972; Cooper et al. 1975; Fogarty et al. 1980; Krouse 1980, 1981; Stasko and Campbell 1980; Ennis 1984). In our studies 91% of the commercial recaptures occurred within the study area (Fig. 10, < 8 km from MNPS). Of the lobsters recaptured outside the study area, most moved to the east (97%; Fig. 11). Lund et al. (1973) reported similar results for tagging studies in LIS. A few lobsters ($n = 13$) traveled considerable distances where they were caught on the edge of the continental shelf (Figure 12).

Figure 10. Locations and numbers of tags returned by lobstermen in the Millstone area. An additional 3,014 tags returned with inaccurate recapture points were reported as being caught around Millstone Point.

Figure 11. Locations and numbers of tags returned by lobstermen in Long Island and Block Island Sounds.

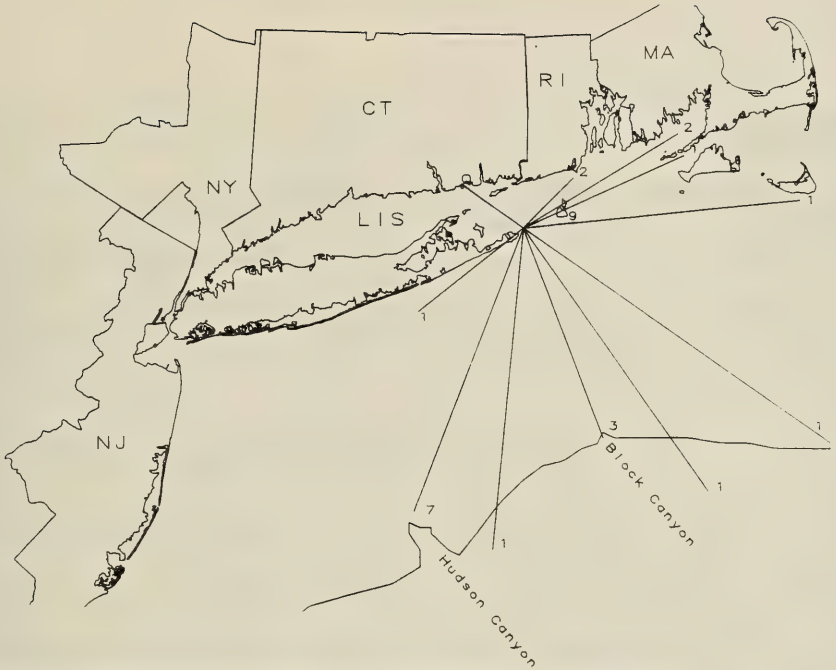

Figure 12. Locations and numbers of tags returned from outside of Long Island Sound.

Other researchers working in waters off New England and on the continental shelf demonstrated similar exchanges between the inshore and offshore populations (Saila and Flowers 1968; Uzmann et al. 1977; Fogarty et al. 1980). Judging from the number of returns from the Race area, it is believed that these lobsters leave the Sound through that deep water channel southwest of Fishers Island. Once out of the Sound lobsters would have moved southerly toward the deep water canyons (Block and Hudson) or easterly (Point Judith, RI, Buzzards Bay, MA, Martha's Vineyard and Nantucket Sounds). Recent work by Campbell and Stasko (1985) off southwestern Nova Scotia revealed that mature male and female lobsters (> 95 mm CL) moved greater distances than immature lobsters (< 95 mm CL). This hypothesis is difficult to assess from our study results since exploitation rates are very high and most of the legal lobsters tagged are caught by lobstermen shortly after release. However, 6 of the 7 lobsters caught at Hudson Canyon were legal-sized and at Block Canyon all 3 were legal-sized.

Entrainment

Lobster larvae were found in ichthyoplankton (IP) samples of the cooling water at Units 1 and 2 discharges from 1977 to 1985. Because lobster larvae distribution in the water column is patchy, large volumes of water must be sampled to collect them. Thus, beginning in 1984, we initiated a special lobster larvae entrainment study, these samples filter much larger volumes of cooling water (4000 m^3) than do the IP samples (400 m^3). This sampling effort, focusing on lobster larvae in the cooling waters was initiated in anticipation of Unit 3 start-up.

The timing of lobster larvae collected in entrainment samples corresponded to the developmental stage of egg masses carried by berried females. In May, the number of berried females collected in traps was high and the development of the eggs indicated that hatching was imminent. In June, the number of berried females carrying ripe eggs declined and by July the low number of berried females caught indicated the completion of the biennial spawning cycle. Although the hatching process and stage duration is temperature dependent (Templeman 1936) and most intense at temperatures of about 20 °C (Hughes and Mattheissen 1962), the larval phase is completed in 25-35 d under normal conditions. In general, since 1977, larvae were collected from mid-May through late-June.

Scarratt (1964, 1973) provided estimates of lobster larvae survival between stage I and IV for a Canadian lobster population. In our study, most of the larvae collected were Stage I and presumably hatched nearby (Table 14; 88% in 1984; 87% in 1985).

Table 14. Summary of 1984 and 1985 lobster larvae entrainment studies.

	1984			1985		
	Day	Night	Total	Day	Night	Total
No. samples collected:	64	58	122	46	47	93
No. samples with larvae:	11	16	27	11	23	34
			Number of Larvae			
Stage I	15	73	88	56	69	125
Stage II	0	1	1	0	1	1
Stage III	0	1	1	3	2	5
Stage IV	1	11	12	2	10	12
Total	16	86	102	61	82	143

Larval survival was not calculated due to the low occurence of Stage II and III larvae relative to Stage IV which may have been an influx from more distant hatching. Lund and Stewart (1970) indicated that larvae produced from the western LIS population are responsible for the gradual recruitment of fourth and fifth stage lobsters in middle and eastern LIS. Their hypothesis appears to be true given the number of fourth stage larvae collected in our entrainment studies.

Survival of lobster larvae after passing through the plant's cooling water system was observed in both entrainment study years. In 1984, 2 stage IV, and in 1985, 2 stage I and 1 stage IV survived after passing through the plant indicating that entrainment mortality is lower than the assumed 100%. Similar findings at other power stations have been reported. Collings et al. (1981) reported 14% survival for lobster larvae (Stage II) collected at the Canal Electric Company, Sandwich, MA.

In both of the lobster larvae entrainment studies (1984-85), and the IP samples collected since 1977, more larvae were collected in night samples than in day samples (Table 14). While this may appear at first to contradict the photo positive behavior of lobster larvae observed by many researchers (Fogarty 1983), a combination of factors may explain the contradiction. Diurnal vertical distribution is apparently related to light intensity, and larvae tend to disperse from surface waters during night except under bright moonlight (Templeman 1939). The fact that more larvae were collected at night when surface densities have been reported to be lowest may result from a combination of this lobster larvae behavior and the intake structure design. The intake structures have curtain walls which extend down into the water column about 2 m below MLW. This means that cooling water is drawn from well below the surface. Therefore, since lobster larvae disperse from surface waters during darkness, they are more susceptible to entrainment at night regardless of tidal stage.

Two 24 h samplings were conducted in 1985 to substantiate our initial findings of higher larval densities in night samples and to determine peak diurnal abundance of lobster larvae in the cooling waters. The numbers of larvae found on the two sampling dates were not significantly different; 19 larvae were collected on 28 May and 17 were collected on 4 June (Table 15).

Table 15. Summary of 1985 24-hour lobster larvae entrainment studies.

Time of Day	28 May no. larvae collected	4 June no. larvae collected
0800	--	0
0900	0	0
1000	0	0
1100	0 Low 1120	0 High 1104
1200	0	1
1300	0	0
1400	0	1
1500	0	0
1600	0	0
1700	0 High 1719	1 Low 1709
1800	0	0
1900	0	0
2000	1[a]	1
2100	3	0
2200	3	2[b]
2300	1 Low 2357	4 High 2316
2400	0	5
0100	3	0
0200	1	1
0300	0	1
0400	1	0
0500	2 High 0540	0
0600	4	0 Low 0605
0700	0	0
0800	0	--
Total	19	17

[a] One larva collected alive.

[b] One stage II.

The numbers of larvae collected in night samples (27) was significantly higher than the numbers collected in day samples (9) confirming our 1984 observations. The two 24 h collections were conducted 1 wk apart to examine the effect of tide on lobster larvae entrainment under different day-night conditions. Results indicated that lobster larvae entrainment was not significantly influenced by tidal stage.

Lobster larvae entrainment estimates from our IP program 1977-85 and from the 1984-85 lobster larvae studies are presented in Table 16. The estimates were calculated by summing the volumes of all samples collected over the study period, dividing that volume into the total cooling water volume over the same period and multiplying that proportion by the number of larvae collected over the hatching

season. The low numbers of larvae collected, and the low entrainment estimate for IP entrainment samples in 1984-85 compared to the special lobster larvae samples in those years, supports the need to apply the new sampling methodology for the assessment of lobster larvae entrainment losses under 3-unit operating conditions, given the reductions in IP sampling effort in 1983 and 1985.

Table 16. Summary of entrainment estimates for lobster larvae collected in ichthyoplankton samples 1977-85 and in 1984-85 lobster larvae entrainment studies.

Year	Dates Found	Number Collected (number samples)	Total Volume (m^3) of samples collected May-Aug ($x10^6$)	Total U-1 + U-2[a] Volume (m^3) May-Aug ($x10^6$)	Entrainment estimate May-Aug
			Ichthyoplankton Samples		
1977	8Jun-6Jul	19 (10)	0.125	517.9	78,721
1978	5Jun-10Aug	74 (24)	0.121	662.9	405,410
1979	11Jun-11Jul	60 (17)	0.151	467.5	185,762
1980	29May-3Jul	37 (14)	0.133	534.8	148,780
1981	27May-13Jul	18 (13)	0.138	556.3	72,561
1982	1Jun-28Jul	45 (26)	0.128	677.9	238,324
1983	31May-20Jul	9 (8)	0.060	516.0	77,400
1984	22May-19Jun	2 (2)	0.060	538.0	17,933
1985	15May-10Jul	4 (4)	0.036[b]	388.7[c]	43,189[c]
			Lobster Larvae Samples		
1984	21May-10Jul	102 (27)	0.505	538.0	108,665
1985	15May-29Jul	143 (34)	0.371	388.7[c]	149,822[c]

[a] U-1 = Millstone Station Unit 1; U-2 = Millstone Station Unit 2

[b] Total volume (m^3) of entrainment samples sorted for lobster larvae from May through July.

[c] May through July.

Impingement

Annual impingement estimates for lobsters collected on Unit 1 and 2 traveling screens from 1975 to 1985 are presented in Table 17. In general, impingement of lobsters is highest during the summer months and coincides with peak catch in traps (NUSCo 1982, 1983, 1984, 1985, 1986). The number of lobsters impinged at Units 1 and 2 was highest in 1982, corresponding with the highest annual catch. Impingement of lobsters and other species is also closely related to plant operations. When units are down for scheduled refueling or maintenance, cooling water demands are considerably less than at full power. Thus the disparity in impingement estimates between units and over years and the lack of correspondence with trap catch values with the exception of 1982 is related to cooling water demands.

Table 17. Annual impingement estimates [a] for lobster collected at Units 1 and 2 from 1975 to 1985.

	Unit 1	Unit 2	Both Units
1975	734	56	790
1976	479	663	1142
1977	240	310	550
1978	245	261	506
1979	323	426	749
1980	368	405	773
1981	665	1009	1674
1982	938	1041	1979
1983	999	497	1496
1984	b	1220	1220
1985	b	480	480
Total	4991	6368	11359

[a] Values for the 1975-76 estimates are based on 7 days of sampling per week. The 1977-83 values are based on 3 days of sampling per week and are extrapolated based on flow rates to represent the estimated total number impinged.

[b] Unit 1 sluiceway began operating December 1983.

The mean sizes of lobsters impinged during the period 1975-85 have ranged from 48.6 to 64.9 mm CL and were smaller than the trap catch values (NUSCo 1982, 1983, 1984, 1985, 1986). Smaller lobsters may enter the screen house through the course bar racks more readily than larger lobsters which are seldom impinged. Since 1982, when we began recording the sex of impinged lobsters, male to female ratios ranged from 1.0:0.47 to 1.0:0.58 and were similar to those reported for the inshore Jordan Cove and Intake stations (NUSCo 1982, 1983, 1984, 1985, 1986). The percentage of culled lobsters was always greater in impingement samples (30-50%) when compared to trap catch values (wood 9-17%; wire 10-16%) and is probably related to the high pressure (80 psi) wash used to remove debris from the screens (NUSCo 1982, 1983, 1984, 1985, 1986).

A fish return system (sluiceway) was constructed at Unit 1 and began operating in December 1983. Prior to the sluiceway operating at Unit 1, and presently at Unit 2, 100% mortality occurs to organisms impinged at MNPS on non-impingement sampling days. On those days that impingement counts were made, survival of lobsters impinged at Units 1 (1975-83) and 2 (1975-1985) ranged from 64-80% (NUSCo 1982, 1983, 1984, 1985, 1986). Generally, highest mortality of lobsters occurred during the peak molting period when lobsters are soft and easily damaged and during the later summer months when water temperatures are highest (NUSCo 1982, 1983, 1984, 1985, 1986). The design and operation of the Unit 3 traveling screens should improve survival and minimize damage to lobsters associated with the impingement process at MNPS. A low pressure (10 psi) screen wash will be used to remove organisms from the screens. The organisms are carried from a fish trough to a sluiceway which returns the organisms back to Niantic Bay.

SUMMARY

1. Annual total CPUE from 1976 to 1985 ranged from 0.56 to 2.10 lobsters per trap. Total CPUE was significantly higher for wire than wood pots. However, the CPUE of legal lobsters (> 81 mm CL) was similar for wood and wire pots. During 1985, low CPUE values at the Intake were attributed to dredging activities in the vicinity of the intake structures.

2. Size frequency distributions indicated that wire pots caught significantly more small lobsters (< 75 mm) than did wood pots. Since wire pots have been used, annual mean CL's have been consistent (range 70.8-71.8). A strong prerecruit size class was observed during 1982 and resulted in record landings throughout LIS in 1983 and 1984.

3. Male to female sex ratios of lobsters were close to 1:1. However, the Twotree station, 1.5 km offshore, has yielded consistently higher numbers of females, when the three sampling stations were compared.

4. The size frequency distributions of berried females and abdomen width/carapace length ratios of females indicated that female lobsters attain sexual maturity in this area at about 50 mm. Twotree had consistently higher proportions of berried females of the three stations.

5. The frequency of molted lobsters in the catch varied over years. Although peaks occurred in early summer, in some years fall molting peaks were also observed. Growth per molt averaged 14.1% for males and 13.7% for females.

6. The percentage of culls ranged from 9.0 to 17.4%; more lobsters collected in wood pots experienced claw loss (14.4%) than lobsters caught in wire pots (12.7%).

7. Since 1975, we tagged 47,259 lobsters and subsequently recaptured 8,053 (17%). Commercial lobstermen recaptured 13,394 (28%) of the tagged lobsters released in our area.

8. Our tagging studies indicate lobster movements are mostly restricted to the local area since 91% of the commercial recaptures were made within the study area. However, some lobsters moved more

than 100 km out of LIS, and were caught on the edge of the continental shelf (Block and Hudson canyons).

9. Lobster larvae entrainment studies conducted in 1984-85 provided better estimates of entrainment losses than estimates made from ichthyoplankton samples. More lobster larvae were collected in night samples than in day samples. Higher numbers of lobster larvae in night samples were related to larval behavior and intake structure design.

10. Since 1975, an estimated 11,359 lobsters were caught on the intake traveling screens at MNPS. Since it began operating at Unit 1, a fish return system improved the overall survival of impinged lobsters.

CONCLUSION

Results from our studies indicated that the local lobster population is highly exploited; more than 90% of legal lobsters are removed by fishing. The commercial and recreational catches were highly dependent on the number of lobsters in the prerecruit size class. Because lobsters require at least 4 years of growth before they are vulnerable to our traps, and an additional 2-3 years to reach marketable size, there is a lag of about 6 years between the time of a potential impact on larvae and the time at which we can detect that impact. Therefore, plant induced impacts or lack thereof on larval stages that may have occurred since 1975 might have been observed in the adults caught in our traps beginning in 1981-82; yet, thus far, the data do not indicate such a stress. Dredging activities in the vicinity of the intakes displaced lobsters from that area; however, lobsters are expected to return soon after the sediments have stabilized.

The sensitivity of our program in defining population trends (i.e., observing the strong prerecruit class in 1982) and impacts (displacement of lobsters as a result of dredging) is vital to our evaluation of impacts associated with the operation of three units at Millstone Point. If changes occur in the local lobster population, they will be detected by the alteration of the basic population parameters now being collected. The stability of these parameters after the start up of Unit 3 will demonstrate the effects (if any) of MNPS operations on the local lobster population.

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Contents

EXPOSURE PANEL PROGRAM 1
INTRODUCTION 1
EXPOSURE PANEL STUDY 3
MATERIALS AND METHODS 3
DATA ANALYSIS 8
RESULTS 10
Fouling Species 10
Wood-boring species 16
DISCUSSION 23
DISTRIBUTION STUDY OF *TEREDO NAVALIS* AND *TEREDO BARTSCHI* 25
MATERIALS AND METHODS 25
DATA ANALYSIS 27
RESULTS 27
DISCUSSION 29
TIMBER STUDY 30
MATERIALS AND METHODS 30
DATA ANALYSIS 31
RESULTS 32
DISCUSSION 36
SUMMARY 39
CONCLUSIONS 40
REFERENCES CITED 41

EXPOSURE PANEL PROGRAM

INTRODUCTION

The exposure panel program monitors the abundance of fouling and wood-boring organisms in the vicinity of the Millstone Nuclear Power Station (MNPS). Since 1975, when the subtropical shipworm, *Teredo bartschi*, was first collected in the MNPS effluent, the primary focus of this program has been the monitoring of shipworms. Marine woodborers and specifically shipworms are primarily responsible for the decomposition of wood in marine and estuarine waters. In the last 25 years, as industrial warm water effluents increased in number, several investigators reported increased growth and fecundity of shipworms within these thermally enhanced environments (Naylor 1965; Board 1973; Turner 1973; NUSCo 1982). For example, *T. bartschi* and *T. furcifera* caused extensive damage to marinas in the vicinity of the Oyster Creek Nuclear Generating Station, New Jersey in 1971 (Turner 1973; Hoagland and Turner 1980; Hoagland 1981). *Teredo bartschi* primarily occurs south of Cape Fear, North Carolina and in the coastal waters of the Gulf of Mexico. Although this species has not been collected beyond the influence of the undiluted effluent at MNPS, we have conducted special studies between 1981 and 1985, and established that *T. bartschi* can survive winter water temperatures in the Millstone Point area and reproduce at local summer temperatures (report in preparation).

Fouling organisms, plants and animals that colonize new substrata, are also important in the exposure panel program, for two reasons. First, some fouling species are sensitive to thermal stresses (Naylor 1965; Cravens 1981) and the survival and growth of their sessile adult stages are influenced by the water quality surrounding the substratum to which they attach (Hillman 1975, 1977). Second, their abundance may affect the recruitment of wood-boring species (Weiss 1948).

Artificial substrata have been used to study community structure and temporal variability of fouling and wood-boring organisms (Turner 1947; Cairns 1982; Manyak 1982). Materials such as glass, plexiglass, ceramic tiles, asbestos-cement and various types of wood have been used as artificial substrata. Although the type of substratum, length of exposure period, and deployment strategies influence the patterns of community colonization (Grave 1928; Zobell and Allen 1935; Schoener 1974; Shafto 1974; Osman 1977;

Sutherland and Karlson 1977), these factors can be standardized to allow comparisons between communities that develop at different sites, or in different seasons. Several environmental monitoring programs have used exposure panels to assess effects of thermal effluents (Cory 1967; Frame 1968; Cory and Nauman 1969; Nauman and Cory 1969; Hillman 1975, 1977; Young and Frame 1976; NAI 1979; Maciolek-Blake et al. 1981; Osman et al. 1981; NUSCo 1982). Our Exposure Panel Program objectives are:

1. to monitor the abundance of marine woodborers at five sites in the Millstone Point area,

2. to quantify the loss of wood associated with the presence of woodborer populations in the vicinity of MNPS,

3. to monitor the dispersal of *Teredo bartschi* in terms of distance from the Millstone Quarry, and,

4. to monitor the abundance of prevalent fouling organisms, and to investigate their relationship to woodborer abundances in the Millstone Point area.

To achieve these objectives, three separate studies were conducted. The first (Exposure Panel Study) used exposure panels to monitor the abundance of fouling and wood-boring species, as well as the associated wood-loss. The second study (Distribution Study of *Teredo navalis* and *Teredo bartschi*) used exposure panels deployed in close proximity to the MNPS discharge to monitor the distribution of shipworms, in relation to the thermal effluent. The third study (Timber Study) used commonly available dock building materials to quantify wood-loss.

The purpose of this report is to provide a summary of results from all Exposure Panel Program studies performed during 2-unit operation. Space limitations required considerable condensation of information; more detailed data are included in Appendix EP I.

EXPOSURE PANEL STUDY

MATERIALS AND METHODS

Fouling and wood-boring organisms have been monitored in the vicinity of Millstone Point since 1968. This report summarizes data collected from 1979 to 1986, following evaluation and modification of the methodology and objectives of the exposure panel program. These changes were based on the recommendations from several studies that critically reviewed our program (Brown and Moore 1977; Battelle 1978a, 1979; NUSCo 1983). The most important changes made were: (1) introduction of replicate panels, (2) reduction of the exposure period from twelve months to six months, and (3) restriction of data collection to the wooden side of each panel. Adding replicates increased the power of our statistical analyses, and reducing the exposure period decreased the likelihood that a panel would be entirely degraded, and lost before collection. Since the major concern in fouling species was to determine their influence on the abundance of woodborers, only data from the community that developed on wooden panels were needed for our analyses.

The present study used sets of six replicated wood panels submerged at five sites, that were grouped as: treatment sites, White Point (WP) and Fox Island (FI), where potential power plant impacts during 3-unit operation may occur; control sites, Black Point (BP) and Giants Neck (GN), located well beyond the area of predicted power plant influence; and one impacted site, Effluent (EF), which was in the Millstone Quarry where panels were exposed to maximum ΔT's. Water temperatures at EF from 1978-1986 averaged 9-10 °C warmer than those recorded at the Intakes of MNPS Units 1 and 2. The Intake temperatures were considered to represent ambient conditions; seawater temperatures at WP and FI did not vary more than 2 °C from ambient (NUSCo 1982). Water temperatures were derived from the Environmental Data Acquisition Network (EDAN), and from continuous strip chart recorders.

Each of six exposure panels consisted of a knot-free pine board (25.4 x 9.5 x 1.9 cm) which had one face covered by plexiglass. Only the uncovered wood side of each panel was processed. Sets of six replicated panels were bolted to a stainless steel rack which was attached to a stainless steel frame at each site (Fig. 2). The rack and frame assemblies deployed at WP, FI, BP, and GN were suspended from docks by ropes in waters not exceeding 2 m in depth; the lower edge of the panels was maintained 0.2 m off

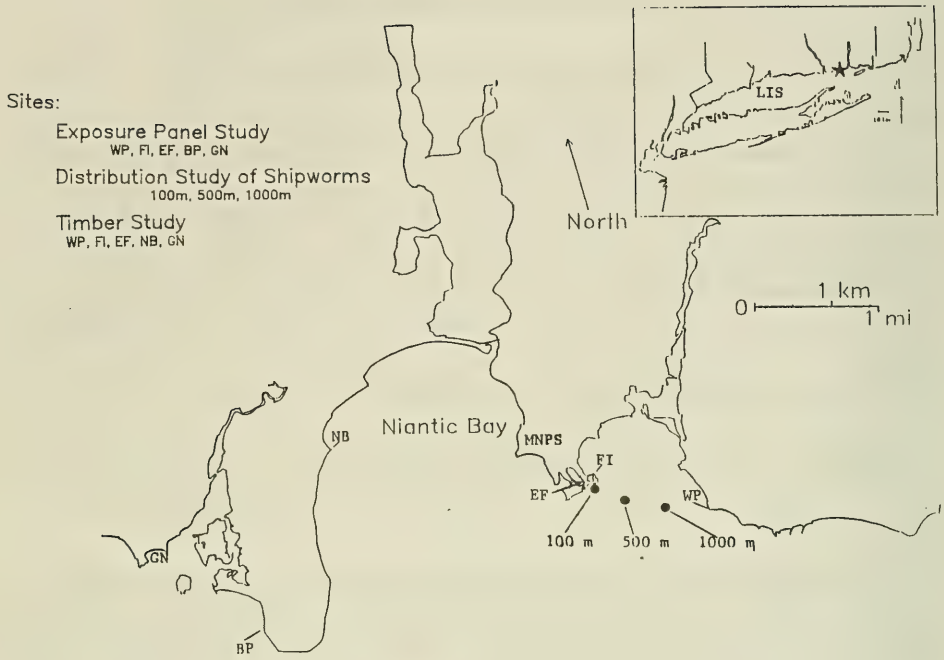

Figure 1. Location of exposure panel and timber sampling sites in the vicinity of the Millstone Nuclear Power Station, (WP = White Point, FI = Fox Island, EF = Effluent, BP = Black Point, NB = Niantic Bay Yacht Club, GN = Giants Neck, 100 m = trawl-line at 100 m, 500 m = trawl-line at 500 m, 1000 m = trawl-line at 1000 m).

the bottom. At EF, two rack and frame assemblies were used; the first 1 m below the water surface, and the second about 1 m off the bottom at a depth of 10 m. This design was adopted to identify organisms that have a settling preference for substrata near the surface, and those that settle near the bottom.

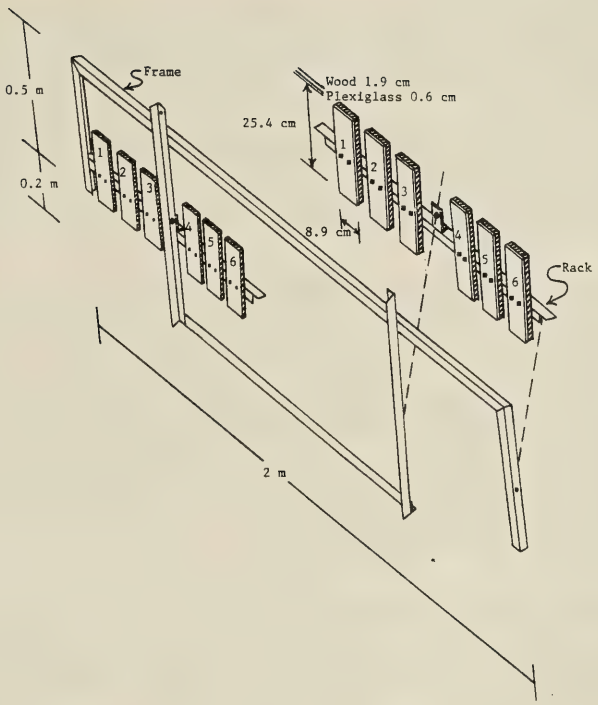

Figure 2. Frame and rack assembly used for holding six-month, six-replicate exposure panels at sites in the vicinity of the Millstone Nuclear Power Station.

The panels were placed at each site in February, May, August and November and were collected six months later in August, November, February and May, respectively. This provided four exposure periods, each overlapping the next by three months. At the start of each exposure period, one rack of panels was removed for processing and a new rack with fresh panels was deployed. Throughout this report the exposure periods will be referred to using the following abbreviations: Aug-Feb, Nov-May, Feb-Aug and May-Nov. Each abbreviation refers to the month of panel deployment followed by the month of panel collection.

The temperature regimes in each of these exposure periods were different (Fig. 3). Aug-Feb began when ambient water temperatures were warmest (> 20 °C) and ended when water temperatures were coldest (< 2 °C). Feb-Aug began when water temperatures were coldest (< 2 °C) and ended when water temperatures were warmest (> 20 °C). Nov-May occurred during the coldest months (average = 5.8 °C) of the year and May-Nov occurred during the warmest months (average = 17.2 °C). Seawater temperatures in this report were based on data collected from 1978-1986 and the monthly averages were based on the interval from the 15th of one month to the 15th of the next.

After collection, panels were either placed in flowing, filtered seawater and processed immediately or frozen and processed at a later time. Primary cover, as a percentage, was estimated for each organism that occupied more than 1% of the panel surface, e.g., barnacles, bryozoans, tunicates and some algae. In addition, primary cover was estimated for other classifications such as freespace, mud and the dead tests of fouling species, to complete the description of total primary cover for each panel. Numerical abundance was determined for barnacles and mussels by counting the individuals on each panel. If the number of individuals per panel exceeded 100, six subsamples of 1 x 1 inch were randomly selected, three from the upper half and three from the lower half of the panel.

The abundance of woodborers was determined after the panel had been scraped of fouling species. All individuals of the genera *Limnoria* and *Chelura* were counted when densities were less than 100 individuals per panel; otherwise, the subsampling scheme previously described for barnacles and mussels was used. Subsampling was always conducted evenly between the top half and the bottom half so that approximately 100 individuals were collected from each panel. After assessing the limnorid and chelurid abundances, panels were frozen and subsequently radiographed using a 250 kV X-ray tube (80 kV, 5 mA, for 1.2 min). The radiographs were used to visually estimate the number of shipworms, *Teredo navalis* and *T. bartschi*, and the percentage of wood lost per panel. The percentage of wood lost was expressed as the average "wood-loss" assigned to the the entire panel by rating the general proportions of bright areas, caused by various densities of shipworm tubes and the dark areas caused by various degrees of wood-loss. To determine the species of shipworms collected, shipworms were randomly removed from the panels until all or at least 100 individuals were identified from each site. Shipworms < 5 mm in length were classified only as juvenile teredinids because their pallets were too small and underdeveloped for making accurate identifications.

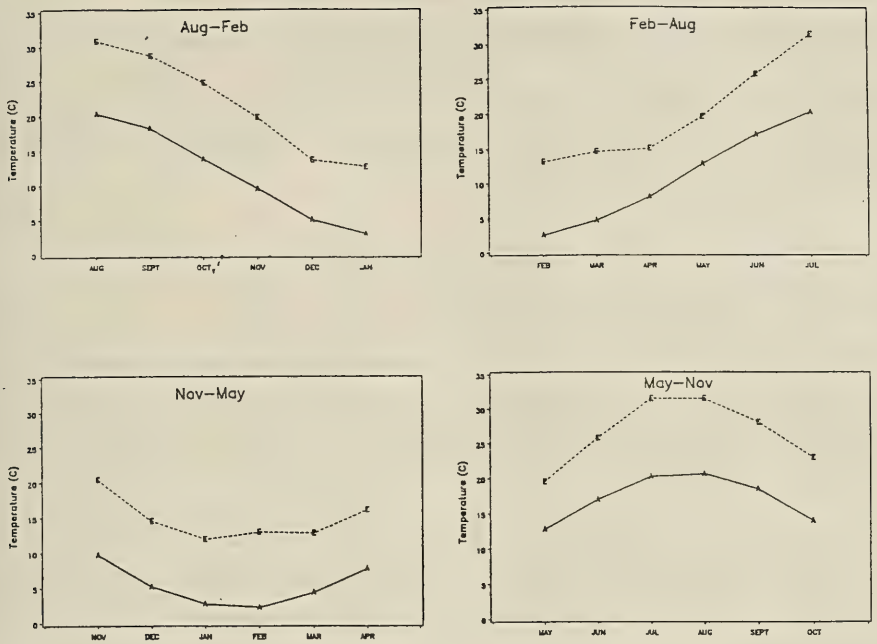

Figure 3. The average monthly seawater temperatures from 1979-1986 during the four six-month exposure periods of the Exposure Panel Program at the Millstone Nuclear Power Station (E = effluent, A = ambient). Note that these monthly averages are from the 15th of one month to the 15th of the next.

In 1985 and 1986, a second estimate of wood-loss was obtained, based on weight. This measure compared the pre-deployment weight of each panel with its weight after retrieval. Panels were dried at 80 °C for 96 h and weighed to the nearest 0.1 g before deployment. After collection and panel processing, each panel was soaked in 10% HCl for at least 48 h to dissolve calcium carbonate tubes secreted by shipworms. Next, the panel was soaked for at least 48 h in two changes of fresh water to rinse the HCL away before the panel was oven dried at 80 °C for 96 h. Panels were weighed to the nearest 0.1 g immediately upon removal from the drying oven.

DATA ANALYSIS

The data represented in this report comprise five collection years prior to three-unit operation at MNPS. The first set of panels was deployed in November 1978 and the last set of panels was collected in May 1986. The collection years summarized in this report are: 1979 with three exposure periods, 1980 and 1981 with four exposure periods each, and more recently 1985 and 1986 with two exposure periods each (Table 1). The monitoring of these panels was suspended from November 1981 to February 1985 (Appendix EP II, U.S. NRC, Docket Nos. 50-245 and 50-336, LS05-81-04-006, Exposure Panels) to conduct special studies concerning the life histories of two shipworms, *Teredo navalis* and *T. bartschi*, in relation to the mixing of effluent and ambient water at MNPS (report in preparation). These studies were undertaken to acquire a better understanding of the limiting factors which restricted *T. bartschi* to the Millstone Quarry.

The four exposure periods in each collection year do not represent a continuum because each period overlaps the next by three months. Therefore, all averages and other summary statistics were computed by exposure period. Data reported as percentages of primary cover were the estimated proportions of exposed wood surface found covered by the most common species of foulers. Since all the exposure panels were of equal size, the percentages of cover were treated as regular (or additive) count-data rather than as ratios. These data for fouling species were based on 197 cm^2 of panel surface, while wood-loss estimates were based on the entire volume of a panel, or 422 cm^3.

Table 1. The number of six-month exposure panels collected from 1979-1986 in the vicinity of the Millstone Nuclear Power Station.

Site	Collection Year	Exposure Periods			
		Aug - Feb	Nov - May	Feb - Aug	May - Nov
WP	1979	--[1]	6	6	6
	1980	6	6	6	6
	1981	6	6	6	6
	1985	--	--	6	5[2]
	1986	6	6	--	--
	Total	18	24	24	23
FI	1979	--	6	6	6
	1980	6	6	6	6
	1981	6	6	6	6
	1985	--	--	--[3]	--[4]
	1986	6	6	--	--
	Total	18	24	18	18
EF	1979	--	6	6	6
	1980	6	6	6	6
	1981	6	6	6	6
	1985	--	--	6	6
	1986	6	6	--	--
	Total	18	24	24	24
BP	1979	--	--	--	--
	1980	--	--	--	--
	1981	--	--	--	--
	1985	--	--	--[3]	--[2]
	1986	3[2]	6	--	--
	Total	3	6	0	0
GN	1979	--	6	6	6
	1980	6	6	6	6
	1981	6	6	6	6
	1985	--	--	6	6
	1986	6	6	--	--
	Total	18	24	24	24

[1] A dash (--), unless otherwise footnoted, indicates that no panels were deployed or collected.
[2] Panels lost during Hurricane Gloria (Sept. 1985).
[3] Panels exposed for three months.
[4] Panels exposed for nine months.

RESULTS

Fouling Species

Primary Cover. The surfaces of the panels were covered with two types of marine fouling, that which was alive and that which had died prior to collection. These two components comprise the total primary cover. At WP, FI and GN total primary covers were largest during Feb-Aug, and at EF during Nov-May. At WP, FI and GN covers were lowest in Aug-Feb, and at EF in Feb-Aug (Fig. 4). The average primary cover on EF panels was 51%, while the combined average cover for the other sites was 29%.

Of the live organisms, 6 plants and 24 animals were considered dominant. The criterion for determining this dominance was whether a species appeared among the five most abundant at any site during any exposure period. After averaging primary covers from 1979-1986 by exposure period, the dominant foulers accounted for 88-100% of the living cover on panels (Table 2). The most dominant organisms at ambient water sites were *Balanus crenatus*, *Codium fragile*, *Cryptosula pallasiana*, *Botryllus schlosseri*, *Laminaria saccharina*, and *B. eburneus*. The most dominant organisms at EF were *B. improvisus* and *Mytilus edulis*.

As a taxonomic group, barnacles represented the most common genus on our exposure panels. *Balanus crenatus*, *B. improvisus*, *B. eburneus* and *Balanus* juveniles accounted for large primary covers throughout the study (Fig. 5). *Balanus amphitrite*, although collected at every site, was generally responsible for less than 1% of the total primary cover and, for this reason, it was not included in Figure 5. In Nov-May collections, these four barnacles contributed from 33% (EF) to 100% (FI) of the live cover at all sites and in Aug-Feb they contributed 76% of the live cover at EF. *Balanus crenatus* was abundant at all sites during Nov-May, the coldest exposure period. *Balanus eburneus* was most dominant at all sites during the warmest exposure period, May-Nov. *Balanus* juveniles were most abundant on panels collected in May and November, indicating that these two months are within the peak setting periods for barnacles in the Millstone Bight.

Numerical Abundance. Six fouling species were monitored by counting the number of individuals attached to our exposure panels: *Balanus amphitrite*, *B. crenatus*, *B. improvisus*, *B. eburneus*, *Balanus* juveniles and *Mytilus edulis* (Table 3). The temporal and spatial trends in numerical abundance were generally similar to those discussed for primary cover data. *Balanus amphitrite* settled on panels in very

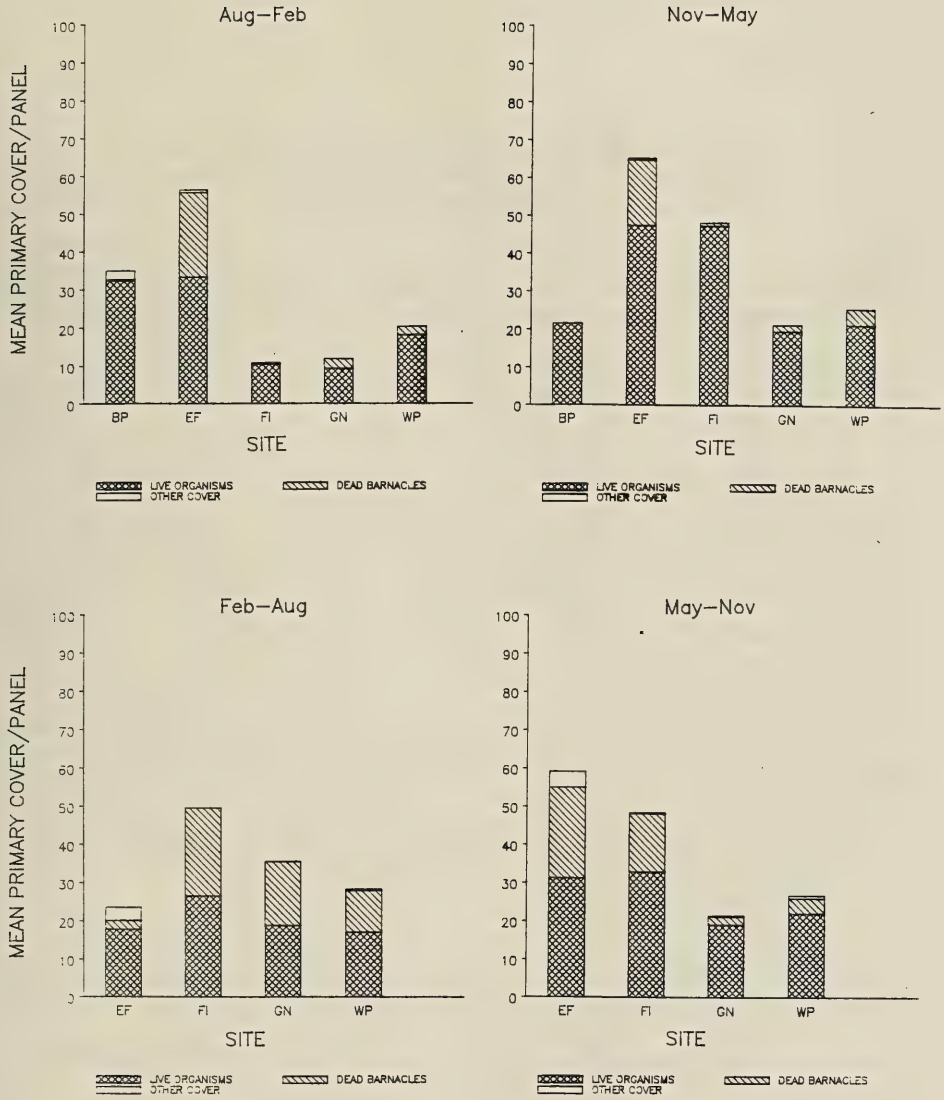

Figure 4. Mean primary covers of attached fouling organisms on wood panels during each of the four six-month exposure periods from 1979-1986 in the vicinity of the Millstone Nuclear Power Station.

Table 2. Dominant taxa from 1979-1986 based on the average percent surface cover of the six-month exposure panels collected in the vicinity of the Millstone Nuclear Power Station. T = trace or less than 0.05% cover.

RANK	TAXA	SITE					TOTAL
		WP	FI	EF	BP	GN	% COVER
AUG - FEB							
1	*Codium fragile*	T	T	--	**27.3**	--	27.3
2	*Balanus improvisus*	1.2[1]	0.1	**21.6**	0.1	0.2	23.2
3	*Botryllus schlosseri*	**12.0**[2]	1.1	--	2.4	**4.8**	20.3
4	*Cryptosula pallasiana*	2.4	**6.0**	0.1	1.6	0.3	10.4
5	*Balanus eburneus*	T	T	2.7	--	--	2.7
6	*Botrylloides leachii*	--	--	2.5	--	--	2.5
7	*Bugula* spp.	T	1.6	0.3	--	0.5	2.4
8	*Balanus* juveniles (< 2mm)	0.1	0.2	1.0	T	0.5	1.9
9	*Mytilus edulis*	0.1	T	1.6	--	T	1.6
10	*Tubularia crocea*	T	--	1.5	--	--	1.5
11	*Balanus amphitrite*	0.7	--	0.5	0.1	0.1	1.4
12	*Molgula* spp.	0.2	T	--	--	1.1	1.3
13	*Halecium* spp.	1.2	--	--	--	T	1.2
14	Hydrozoa	--	0.7	--	--	--	0.7
15	*Serpulid* tubes	T	T	0.3	0.4	T	0.7
16	*Barensia* spp.	--	0.3	0.3	--	--	0.6
17	*Alcyonidium* spp.	--	--	--	--	0.6	0.6
18	*Ceramium rubrum*	T	T	--	0.4	T	0.4
19	*Nicloea venustula*	--	--	--	--	0.3	0.3
	Total % cover	17.9	10.0	32.4	32.3	8.4	20.2[3]
NOV - MAY							
1	*Balanus crenatus*	**10.8**	**35.2**	10.7	6.7	**11.2**	74.6
2	*Balanus* juveniles (< 2mm)	4.5	12.1	1.8	4.0	6.2	28.6
3	*Mytilus edulis*	0.2	T	**17.7**	0.1	T	18.0
4	*Laminaria saccharina*	5.8	0.2	--	**8.8**	2.1	16.9
5	*Alcyonidium* spp.	--	--	9.2	--	--	9.2
6	*Tubularia crocea*	--	--	3.8	--	--	3.8
7	*Balanus improvisus*	--	T	1.8	2.0	--	3.8
8	*Obelia* spp.	0.3	0.1	--	--	--	0.4
9	*Punctaria plantaginea*	--	T	--	--	--	T
10	Bacillariophyceae	T	T	--	--	T	T
	Total % cover	21.6	47.6	45.0	21.6	19.5	31.1

[1] Underlined percent covers indicate that the taxon was one of the top five dominants for that site and exposure period.

[2] Bold-face percent covers indicate that the taxon was the most dominant for that site and exposure period.

[3] This is an average, i.e, sum of the total % covers divided by the number of sites.

Table 2. (cont'd)

RANK	TAXA	SITE					TOTAL % COVER
		WP	FI	EF	BP	GN	
FEB - AUG							
1	*Balanus crenatus*	**6.6**	**14.3**	0.2		**13.9**	35.0
2	*Balanus improvisus*	1.2	0.9	**9.9**		0.2	12.2
3	*Cryptosula pallasiana*	1.8	6.0	0.4		0.2	8.4
4	*Balanus eburneus*	2.8	0.3	3.5		0.6	7.2
5	*Botryllus schlosseri*	1.3	0.2	--		2.4	3.9
6	*Ralfsia verrucosa*	T	3.6	--		0.2	3.8
7	*Balanus* juveniles (< 2mm)	0.7	0.3	1.6		0.5	3.1
8	*Bugula* spp.	1.6	0.3	T		0.2	2.1
9	*Balanus amphitrite*	0.9	0.3	0.1		0.6	1.9
10	*Metridium senile*	--	--	1.5		--	1.5
11	*Halichondria* spp.	T	0.6	--		0.1	0.7
12	*Tubularia crocea*	--	--	0.5		--	0.5
	Total % cover	16.9	26.8	17.7		18.9	20.1
MAY - NOV							
1	*Cryptosula pallasiana*	**7.7**	**23.7**	--		3.4	34.8
2	*Balanus eburneus*	5.9	1.0	9.2		**5.1**	21.2
3	*Balanus* juveniles (< 2mm)	2.2	1.3	10.3		3.2	17.0
4	*Balanus improvisus*	1.2	2.1	**11.8**		1.5	16.6
5	*Halichondria* spp.	1.3	0.7	--		1.0	3.0
6	*Bugula* spp.	0.9	1.1	T		0.7	2.7
7	*Balanus amphitrite*	0.1	0.5	T		1.7	2.3
8	*Schizoporella unicornis*	1.4	--	--		0.3	1.7
9	Serpulid tubes	T	0.1	0.1		T	0.2
10	*Crepidula plana*	T	--	0.1		T	0.1
	Total % cover	20.7	30.5	31.5		16.9	24.9

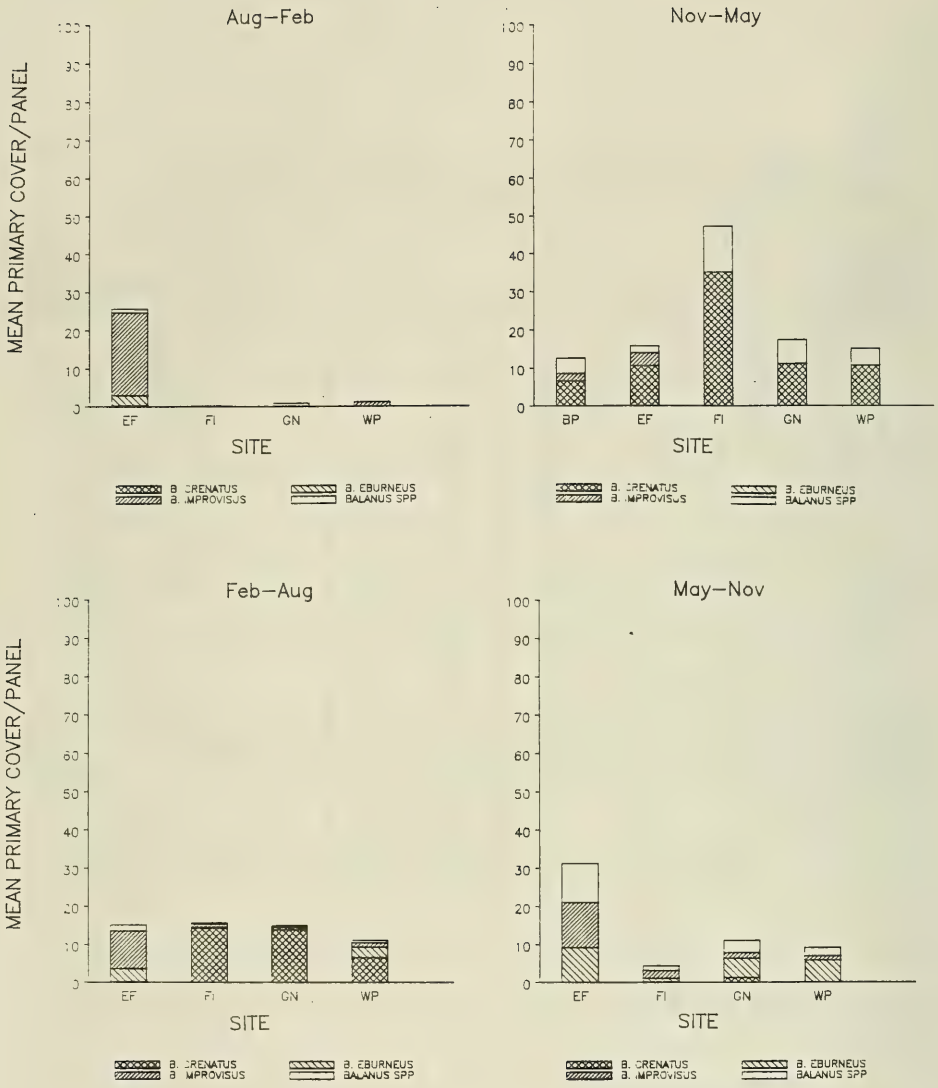

Figure 5. Mean primary covers of four subtidal barnacles on wood panels during each of the four six-month exposure periods from 1979-1986 in the vicinity of the Millstone Nuclear Power Station.

Table 3. The mean numerical abundance of six fouling species on six-month exposure panels collected in the vicinity of the Millstone Nuclear Power Station, from 1979-1986.

TAXA	SITE	MEAN COUNT (±2 standard errors) Aug - Feb	Nov - May	Feb - Aug	May - Nov
Balanus amphitrite	W P	8.6 ± 9.1	0.8 ± 1.7	12.2 ± 10.0	0.3 ± 0.5
	F I	0 ± 0	0 ± 0	1.9 ± 1.6	0.1 ± 0.1
	E F	9.8 ± 9.1	0 ± 0	0.7 ± 0.7	0 ± 0
	B P	1.3 ± 1.3	0 ± 0	---	---
	G N	0.7 ± 0.7	0 ± 0	9.5 ± 8.9	0 ± 0
Balanus crenatus	W P	0.1 ± 0.1	113.5 ± 58.4	61.6 ± 29.3	0 ± 0
	F I	2.0 ± 2.8	350.9 ± 187.2	8.7 ± 7.0	0.4 ± 0.5
	E F	5.2 ± 4.6	27.4 ± 13.6	0 ± 0	0 ± 0
	B P	0 ± 0	280.5 ± 104.5	---	---
	G N	0.1 ± 0.1	247.8 ± 188.6	83.6 ± 47.4	0.9 ± 1.3
Balanus eburneus	W P	0.8 ± 1.7	0 ± 0	21.8 ± 19.2	8.2 ± 5.6
	F I	0.1 ± 0.1	0 ± 0	0.3 ± 0.7	0.4 ± 0.4
	E F	7.4 ± 6.1	0 ± 0	7.5 ± 3.7	18.5 ± 9.3
	B P	0 ± 0	0 ± 0	---	---
	G N	0 ± 0	0 ± 0	3.0 ± 2.6	6.0 ± 4.4
Balanus improvisus	W P	31.5 ± 39.1	< 0.1 ± 0.1	18.5 ± 15.2	4.1 ± 2.9
	F I	5.0 ± 3.2	0.1 ± 0.2	1.2 ± 1.3	0 ± 0
	E F	223.7 ± 152.6	18.6 ± 12.9	59.0 ± 25.1	64.5 ± 40.8
	B P	2.3 ± 1.8	0 ± 0	---	---
	G N	1.2 ± 1.0	0 ± 0	4.5 ± 5.2	3.0 ± 2.4
Balanus juveniles (< 2mm)	W P	12.4 ± 12.2	522.3 ± 233.8	237.5 ± 177.8	10.8 ± 6.7
	F I	62.2 ± 36.0	826.9 ± 448.4	8.2 ± 4.3	9.5 ± 6.7
	E F	53.6 ± 44.1	9.2 ± 2.6	49.5 ± 25.6	118.7 ± 84.7
	B P	0.7 ± 1.3	820.3 ± 287.2	---	---
	G N	92.4 ± 77.0	677.2 ± 309.2	39.8 ± 19.4	18.8 ± 11.1
Mytilus edulis	W P	149.3 ± 110.1	27.9 ± 19.5	7.1 ± 6.9	1.5 ± 2.8
	F I	1.4 ± 1.6	5.6 ± 3.2	0.3 ± 0.3	0 + 0
	E F	167.9 ± 134.6	128.4 ± 74.1	0 ± 0	0 ± 0
	B P	0 ± 0	21.8 ± 8.0	---	---
	G N	3.9 ± 3.5	3.0 ± 2.3	0.5 ± 0.3	0 ± 0

low numbers. The maximum recruitment for this species was 12 individuals per panel during Feb-Aug. *Balanus crenatus* settled in larger numbers than any of the other four barnacles and did so during the coldest period of the year, Nov-May. *Balanus eburneus* consistently settled in low numbers on panels at all sites during Feb-Aug and May-Nov. This species was also a dominant fouler at EF in Aug-Feb. *Balanus improvisus*, like *B. eburneus*, was most commonly collected at EF and had 3 to 19 times more larvae settling and surviving at EF than at any other site. Juvenile barnacles, which represent the pre-adults of the above four barnacles, had strong recruitment onto panel surfaces throughout the study. The largest numbers of juveniles settled during Nov-May, when *B. crenatus* was most abundant. Peak settlement of *Mytilus edulis* occurred during Aug-Feb and Nov-May, and recruitment was largest at EF (Table 2). In contrast, *M. edulis* was absent at EF in the Feb-Aug collections, while it was generally present in small numbers at the ambient water sites.

Wood-boring species

Percentage of wood lost. Wood-loss of exposure panels was based on either direct inspection of each panel (1979) or the visual assessment of panel radiographs (1980-1986). These types of estimates were not as quantitative as that based on the actual weight loss of panels, but the precision of visual method as compared to the weight method was very good ($R^2 = 0.98$). Using a second-order polynomial regression model (Fig. 6), the following equation was provided to predict the percentage of wood lost by weight (WT) from that estimated by visual assessment of radiographs (V):

$$\%\mathbf{WT}\ loss = 6.668 + 0.4204\,(\%\mathbf{V}\ loss) + 0.0032\,(\%\mathbf{V}\ loss)^2$$

Wood-loss (V) was highest in May-Nov (average > 25%) and was primarily caused by *Teredo navalis* (Fig. 7). Wood-loss during the Feb-Aug exposure period was < 5%, and ca. 10% during Aug-Feb. Wood-loss from Nov-May was negligible, as *T. navalis* did not recruit during this period. The wood-loss caused by *T. bartschi* at EF and juvenile teredinids at all sites did not exceed 1%. Among stations, wood-loss has been greatest at GN and least at EF since 1979. At WP and FI, wood-loss has been similar during all exposure periods except May-Nov, when wood-loss was nearly three times larger at WP than at FI.

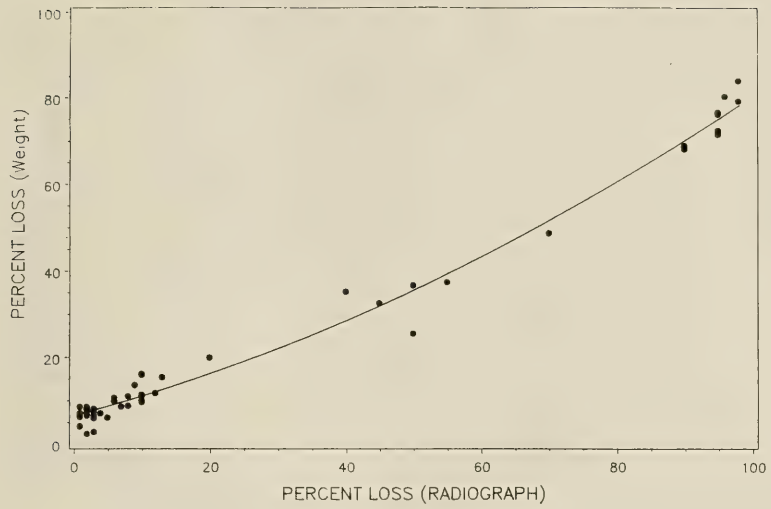

Figure 6. Comparison of wood-loss estimates from exposure panel weights versus visual assessments of radiographs from 1985-1986. These data were collected as part of the Exposure Panel Program at the Millstone Nuclear Power Station.

Numerical abundance. The Millstone area has two general groups of marine woodborers: crustaceans and molluscs (Table 4). The crustaceans are represented in our data by two native taxa: *Limnoria* spp., isopods, and *Chelura terebrans*, an amphipod. Because of the difficulties in identifying *Limnoria* to the level of species, these "gribbles" have been lumped together at the generic level. Recruitment of *Limnoria* onto panels was greatest during the Feb-Aug (88-623 individuals/panel) and May-Nov (up to 1670 individuals/panel) exposure periods (Fig. 8). Of the five sites monitored, GN and WP supported the greatest populations of these isopods. During the May-Nov exposure period, WP had 5 times more *Limnoria* than any other site and nearly 9 times more *C. terebrans*. This latter species has a commensal relationship with dense populations of limnorids, since the chelurids primarily feed on the thin

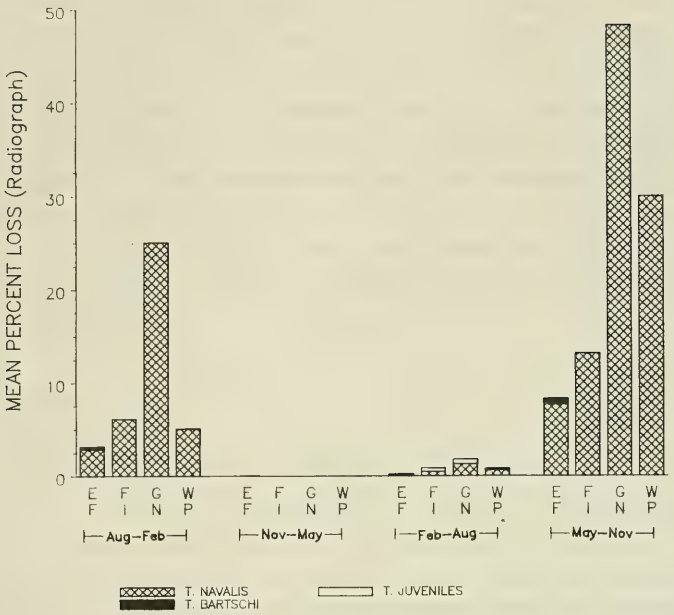

Figure 7. Mean wood-loss caused by shipworms in six-month exposure panels collected from 1979-1986 in the vicinity of the Millstone Nuclear Power Station.

partitions between limnorid tunnels (Barnard 1959). These large densities of wood-boring crustacea at WP during May-Nov were the cause of the increased wood-loss noted earlier at WP for this exposure period.

The molluscan woodborers, shipworms, were represented in the Millstone area by *Teredo navalis*, the native species and by *T. bartschi*, an immigrant species (Table 4). Shipworms were collected from every site during three of four exposure periods (Fig. 9). Juvenile teredinids (< 5 mm in length) representing recent recruitment to panel communities, were most numerous during peak reproductive periods; August

Table 4. The mean numerical abundance of five wood-boring taxa on six-month exposure panels collected in the vicinity of the Millstone Nuclear Power Station, from 1979 -1986.

TAXA	SITE	MEAN COUNT (±2 standard errors) Aug - Feb	Nov - May	Feb - Aug	May - Nov
Limnoria spp.	WP	109.1 ± 52.3	40.7 ± 20.5	581.2 ± 107.6	1670.5 ± 416.4
	FI	38.0 ± 27.7	0.5 ± 0.5	179.3 ± 102.0	164.9 ± 51.6
	EF	1.1 ± 1.5	1.9 ± 1.5	87.8 ± 54.9	6.8 ± 5.9
	BP	3.0 ± 1.2	4.2 ± 2.4	---	---
	GN	201.8 ± 90.6	307.8 ± 147.1	623.2 ± 194.0	341.2 ± 196.8
Chelura terebrans	WP	0.8 ± 0.8	0 ± 0	18.3 ± 14.5	898.0 ± 511.3
	FI	0.9 ± 1.1	0 ± 0	0 ± 0	0 ± 0
	EF	0 ± 0	0 ± 0	0 ± 0	0 ± 0
	BP	0 ± 0	0 ± 0	---	---
	GN	3.1 ± 4.4	0 ± 0	0.9 ± 1.2	100.2 ± 138.3
Teredo navalis	WP	33.3 ± 18.2	0 ± 0	9.4 ± 4.7	50.3 ± 19.4
	FI	9.4 ± 3.9	0 ± 0	7.3 ± 5.0	15.1 ± 4.4
	EF	8.1 ± 4.1	<0.1 ± <0.1	0.5 ± 0.3	7.4 ± 3.3
	BP	6.0 ± 4.2	0 ± 0	---	---
	GN	73.7 ± 44.6	0 ± 0	46.7 ± 21.8	210.4 ± 42.7
Teredo bartschi	WP	0 ± 0	0 ± 0	0 ± 0	0 ± 0
	FI	0 ± 0	0 ± 0	0 ± 0	0 ± 0
	EF	13.4 ± 10.8	0.1 ± 0.1	1.0 ± 0.9	2.3 ± 1.5
	BP	0 ± 0	0 ± 0	---	---
	GN	0 ± 0	0 ± 0	0 ± 0	0 ± 0
Teredo juveniles	WP	0.1 ± 0.1	0 ± 0	23.8 ± 17.1	0 ± 0
	FI	0.1 ± 0.1	0 ± 0	12.6 ± 7.8	0.2 ± 0.4
	EF	0.4 ± 0.4	0.6 ± 0.2	1.0 ± 0.5	6.4 ± 6.4
	BP	0 ± 0	0 ± 0	--	--
	GN	0.1 ± 0.1	0 ± 0	71.6 ± 20.5	0.5 ± 0.7

at ambient water sites and November at EF. Teredinid larvae settling at EF in November were most likely *T. bartschi*, since the major increase in recruitment from November to February was for this species.

Among sites, GN had the greatest recruitment of shipworms (210 shipworms/panel/year), and had 2-4 times as many shipworms per exposure period as any other site, while FI had the fewest (15 shipworms/panel/year) among the ambient water sites. BP, established in 1985, has not been sampled for a long enough period to determine temporal trends. EF was characterized by low abundances of all wood-boring species throughout the study. However, this site supported the only population of *T. bartschi*, which has been collected in the effluent since 1975.

Depth preferences of foulers and woodborers at EF. Two rack and frame assemblies of exposure panels were monitored at EF to determine if fouling species or woodborers had any depth preferences for panels

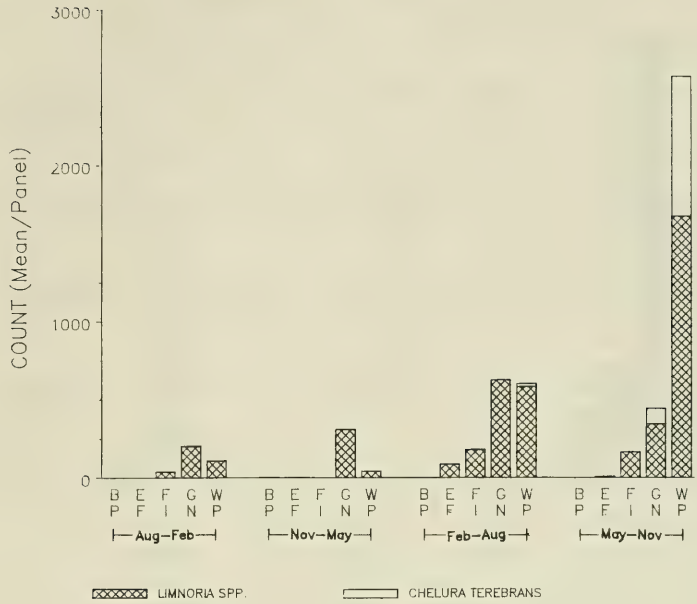

Figure 8. Mean numercial abundance of wood-boring crustacea, *Limnoria* spp. and *Chelura terebrans* in six-month exposure panels collected from 1979-1986 in the vicinity of the Millstone Nuclear Power Station.

placed 1 m below the surface or those placed 1 m above the bottom. Nine taxa were selected for comparison based on their peak occurrences during the four exposure periods sampled from 1985-1986 (Table 5). Of these, four foulers and two woodborers had significantly larger abundances on one set of panels as compared to the other in at least one exposure period. *Alcyonidium* spp., *Balanus eburneus*, *Tubularia crocea* and *Teredo navalis* had their largest abundances on the panels close to the bottom, while *B. improvisus* and *Limnoria* spp. were most abundant near the surface. Even though limnorids were most

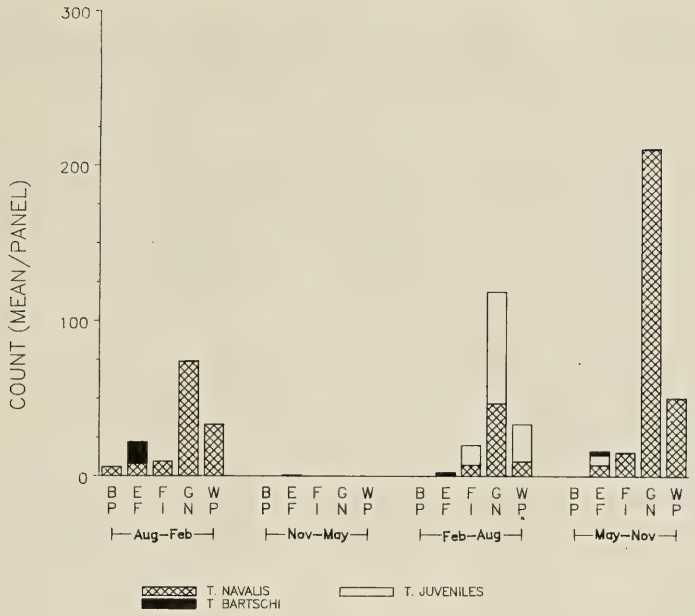

Figure 9. Mean numercial abundance of the wood-boring mollusca, *Teredo navalis*, *Teredo bartschi* and *Teredo* juveniles (< 5 mm in length), in six-month exposure panels collected from 1979-1986 in the vicinity of the Millstone Nuclear Power Station.

abundant near the surface, *T. navalis* was much more destructive. Therefore, general wood-loss in panels was greater near the bottom.

Table 5. A comparison between the recruitment of dominant fouling and wood-boring species on six-month exposure panels suspended at two different depths in the effluent of the Millstone Nuclear Power Station, from February 1985 - May 1986.

Taxa	Exposure period[1]	Mean Abundance Surface	Mean Abundance Bottom	Probability t-test	Probability Wilcoxon 2-sample Test
Alcyonidium spp.	Nov-May	11.8%[2]	17.8%	0.04*	0.07
Balanus eburneus	Feb-Aug	2.2[3]	8.7	0.007*	0.02*
	May-Nov	2.5	9.8	0.008*	0.06
Balanus improvisus	Feb-Aug	9.8	14.8	0.21	0.23
	Aug-Feb	15.0	0.3	0.003*	0.004*
Balanus crenatus	Aug-Feb	13.7	17.7	0.58	0.42
	Nov-May	78.0	147.8	0.15	0.47
Mytilus edulis	Nov-May	70.2	91.0	0.26	0.38
Tubularia crocea	Feb-Aug	0.0%	3.0%	0.01*	0.009*
Wood loss	May-Nov	4.5%	11.7%	0.008*	0.02*
	Aug-Feb	3.0%	0.7%	0.13	0.10
Limnoria spp.	Feb-Aug	307.2	0.5	0.0001*	0.004*
	May-Nov	21.2	1.3	0.07	0.04*
Teredo navalis	May-Nov	0.5	6.5	0.004*	0.004*
	Aug-Feb	0.3	1.5	0.08	0.06
Teredo bartschi	Feb-Aug	4.2	3.3	0.50	0.61
	May-Nov	3.8	7.7	0.19	0.23
	Aug-Feb	2.0	1.7	0.81	0.80

[1] Exposure periods listed are those which had peak recruitment for each species.
[2] #% = average primary cover (n = 6)
[3] # = average numerical abundance (n = 6)
* Difference is significant at least at the 0.05 level.

DISCUSSION

Temporal changes in exposure panel communities are closely associated with seasonal water temperatures and the suites of fouling and wood-boring life stages which are available for settling. In Nov-May, the coldest exposure period, panel communities were dominated by cold and temperate water species: *Balanus crenatus*, *Mytilus edulis* and *Laminaria saccharina*. In May-Nov, the warmest exposure period, these dominant species shifted to the warmer water assemblages of *Cryptosula pallasiana*, *B. eburneus*, *B. improvisus*, *Limnoria* spp., *Chelura terebrans* and *Teredo navalis*. These seasonal trends in recruitment for fouling and wood-boring species have been well documented in other studies (Nair and Saraswathy 1971; Osman 1977, 1978; Sutherland and Karlson 1977; Ibrahim 1981).

The fouling community at EF was different from those at ambient water sites; large primary covers on EF panels were related to accelerated growth in response to elevated temperatures. Cory and Nauman (1969) found that dry weight production of fouling species was on the average 2.8 times greater in the power plant effluent at Chalk Point, Maryland than in its intake waters. Young and Frame (1976) reported that the optimum temperature for growth of *Balanus* spp. was approached more closely during the winter in the Oyster Creek power plant's discharge canal than in the intake canal where growth rate was reduced by cold water.

Conversely, effluent temperatures during the warmest months of the year had negative effects. Settlement and survival of some species were adversely affected by effluent temperatures, since seasonal temperatures exceeded the upper tolerance levels for some of their life stages. Similar exclusion of temperate-boreal species from the Millstone Quarry has been discussed in the Rocky Intertidal section of this report. *Laminaria saccharina*, a temperate-boreal brown alga, has never been collected on panels at EF, yet was a consistent dominant at ambient water sites in Nov-May. Adult stages of *Mytilus edulis* have an upper thermal tolerance of 26-27 °C (Gonzalez and Yevich 1976; Johnson et al. 1983) and the effluent temperatures exceeded 26°C from June to September. Large populations of mussels were observed in the Millstone Quarry during several May collection periods, but totally disappeared during June and July. In addtion, remains of dead barnacles and bryozoans were generally 2.5 times greater at EF than at other sites. This abiotic cover peaked on EF panels during the warmest exposure period of the year (May-Nov).

To date, *Teredo bartschi* has been found only in panels exposed to undiluted effluent, but the destructive potential of this species warranted a series of special studies, conducted from 1981 to 1985 (report in preparation). *Teredo bartschi* is a warm water immigrant, and differs from *T. navalis* in being a long-term brooder instead of a short-term brooder. By brooding its veliger larvae to the pediveliger stage before releasing them, *T. bartschi* pediveligers are able to reinfest wood near their parent populations, which makes this species very destructive to a localized area after it has established a population (Hoagland and Turner 1980; Turner 1973). In contrast, *T. navalis* veligers are released at the straight hinge stage and require at least three weeks of development in the plankton before they metamorphose (Imai et al. 1950; Culliney 1975). *Teredo bartschi* recruitment into EF panels occurred after August with the largest populations occurring in the panels collected in February. However, our investigation concerning the life history of this species has established that *T. bartschi* release pediveliger larvae throughout the year at EF temperatures (report in preparation). This would suggest that the successful recruitment of this species requires temperatures above 22 °C.

Wood-loss data prior to 1979 indicated that the EF site consistently experienced the heaviest annual shipworm infestation of any of the ambient water sites (Battelle 1978b). This contradicted the 1979-1986 data as presented in this report. The reason for this discrepancy is that the location of the exposure panel rack at EF was changed in 1979 from just off the bottom in shallow water to 1 m below the surface in about 10 m of water. The depth preference data described in this report indicates that *T. navalis* occurred in greater abundance 1 m off the bottom than at 1 m from the surface. Others have reported that this species has a strong preference for setting close to the bottom (Grave 1928; Scheltema and Truitt 1956; Turner 1966; Nair and Saraswathy 1971). In fact, its preference is so pronounced that the location of our panels 1 m of the bottom probably missed their peak zone of recruitment. Therefore the 1979-1986 data may underestimate the abundances of woodborers on the bottom. We will evaluate the depth response of *T. navalis* by placing a set of panels in pre-1979 position, i.e., just off the bottom in shallow water.

In summary, the ambient water sites had similar fouling assemblages and trends in abundance from 1979-1986. Total primary cover at all sites was dominated by three barnacle species. *Balanus crenatus* was most dominant at ambient water sites during Aug-Feb and Nov-May, and *B. eburneus* and *B. improvisus* were most dominant during Feb-Aug and May-Nov. Other fouling species that consistently colonized over 5% of the panel surfaces at ambient sites were *Cryptosula pallasiana*, *Botryllus schlosseri* and *Laminaria saccharina*. The primary covers of EF panels were different from those at ambient water sites; *Balanus*

improvisus, *Tubularia crocea* and *Mytilus edulis*, and *Alcyonidium* spp. had consistently large primary covers, while *Laminaria saccharina* and *Botryllus schlosseri* were totally absent from EF assemblages.

Woodborer abundances vary from site to site and were dependent on the time of year the exposure panels were exposed. Trends in primary cover did not appear to affect woodborer abundances. At ambient water sites, *Teredo navalis* was the only shipworm collected. Woodboring crustacea, *Limnoria* spp. and *Chelura terebrans*, were most abundant at WP and GN. The EF site had the only population of *T. bartschi* and this species appeared to recruit to panels only when effluent temperature exceeded 22 °C. *Teredo navalis* colonized panels at EF later in the year than at ambient water sites.

Patterns of abundance and distribution of woodborers and other fouling organisms at the ambient water sites were consistent and predictable from year to year, as were differences between the ambient water communities and those that developed in undiluted effluent. Characteristics of the EF community included enhanced primary cover, temporal shifts in peak abundance of individual species and total primary cover, absence of cold water species, and the unique occurrence of a warm water shipworm, *Teredo bartschi*. Further investigations of the factors that control the distribution of *T. bartschi* are detailed in the next section.

DISTRIBUTION STUDY OF *TEREDO NAVALIS* AND *TEREDO BARTSCHI*

MATERIALS AND METHODS

This study used panels located at 100, 500, and 1000 m from the cuts in the Millstone Quarry through which the MNPS effluent flows into Long Island Sound (Fig. 1). In May of each year a total of 15 knot-free pine exposure panels without plexiglass backers were placed in three wire lobster pots attached to each of three trawl-lines (Fig. 10). In November, nine panels from each trawl-line were collected, leaving six panels to provide substratum for a stock population. Three panels were taken from each lobster pot in a trawl-line and three new ones were added. In May of the following year all 15 panels were collected and replaced with new ones. To date, only two exposure periods have been completed.

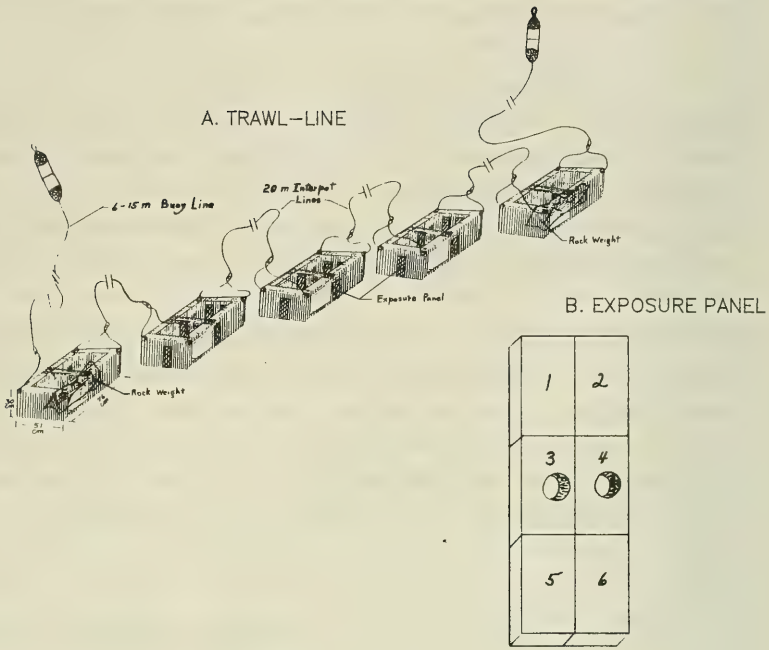

Figure 10. Diagram of an exposure panel trawl-line used to sample the distribution of shipworms in relation to the effluent discharge point at the Millstone Nuclear Power Station (a. trawl-line of five lobster pots with the locations of the 15 pine panels; b. pine panel showing the sections used for subsampling).

The panels collected in November were exposed during the period of peak recruitment for shipworms. Therefore, only six of the nine panels collected from each trawl-line were selected for shipworm identification. Each panel was cut into six equal parts, and all shipworms in one of the parts were removed and identified. Because the parts chosen came from different locations in each panel (Fig. 10), the six parts processed

were, in fact, a "composite" of an entire panel. The same sampling scheme was used for each of the three trawl-lines.

Shipworm infestation was assumed to be minimal in panels exposed from November to May. Hence, all 15 panels were radiographed using the X-ray method described earlier, and if shipworms were present, they were removed and identified.

DATA ANALYSIS

Data analyses consisted of comparisons between sites using observations made on panel sections paired by their location in the panel. This type of pairing was necessary, because shipworm recruitment rates vary with the location of the panel section observed (e.g., shipworm infestation is greatest on the bottom and top ends of a panel). Parametric t-tests and non-parametric Wilcoxon 2-sample tests were used to compare the occurrences of *Teredo navalis* between pairs of sites.

RESULTS

A total of 18 panels were collected in November 1985, after a six month exposure at distances of 100, 500 and 1000 m from the Millstone Quarry. Over 1500 shipworms were removed and identified from these panels (Table 6). The estimated total recruitment of shipworms was 615 per panel at 100 m, 526 at 500 m and 388 at 1000 m. *Teredo bartschi* were collected only in panels at 100 m, and they represented 2.3% of the total number of shipworms which were collected at that distance. However, one of the 14 individuals collected was brooding pediveligers, indicating that *T. bartschi* can reproduce outside the quarry.

Both parametric and non-parametric statistical comparisons were made concerning the abundances of shipworms by panel section and distance. None of the three comparisons among the sites were significant at $\alpha = 0.05$. However, there is a trend of decreasing abundance with increasing distance from the quarry (Table 7). Low number of observations and high degree of intra-site (within site) variability contribute to the lack of statistical significance; more data may substantiate a distinct trend.

In May 1986, all 45 panels, representing all distances, were collected, x-rayed, and shown to be devoid of shipworms.

Table 6. Distribution of shipworms in relation to the effluent discharge point at the Millstone Nuclear Power Station, from May - November 1985.

Distance	Species	Number per section 1	2	3	4	5	6	Total no. per panel	Percent
100 m	*Teredo navalis*	145	116	49	32	161	98	601	97.7
100 m	*Teredo bartschi*	4	0	4	0	0	6	14[a]	2.3
TOTAL								615	
500 m	*Teredo navalis*	85	83	58	98	92	110	526	100.0
500 m	*Teredo bartschi*	0	0	0	0	0	0	0	0.0
TOTAL								526	
1000 m	*Teredo navalis*	98	83	73	43	54	37	388	100.0
1000 m	*Teredo bartschi*	0	0	0	0	0	0	0	0.0
TOTAL								388	

[a] One individual was brooding pediveligers.

Table 7. Statistical comparisons in the mean number of *Teredo navalis* collected in exposure panels, in relation to the effluent discharge point at Millstone Nuclear Power Station, from May - November 1985.

Difference	n	Mean difference	S E	Probability t-test	Probability Wilcoxon 2-sample test
100 vs 500m	6	12.2	21.1	0.59	0.52
100 vs 1000m	6	35.2	19.8	0.14	0.26
500 vs 1000m	6	23.0	15.3	0.19	0.09

DISCUSSION

The panels at 100, 500, and 1000 m represent a series of samples collected along a straight line at increasing distances from the quarry. The direction of this transect was the same as the ebb tide flow of the effluent plume. The data described a consistent decrease in panel recruitment of *Teredo navalis* from 601 at 100 m to 388 at 1000 m. This trend may indicate that recruitment of *T. navalis* is enhanced in the effluent mixing zone, even though the differences in shipworm densities were not statistically different ($p < 0.05$) from 100 to 1000 m.

This was the first study investigating the distribution of *Teredo bartschi* beyond the undiluted effluent or EF site. Panels at the 100 m sampling site were within the effluent mixing zone, and were periodically flushed by effluent waters; the depth at this site was 5 m. Incursion of effluent water permitted the successful recruitment of *T. bartschi* in panels at 100 m. Studies on the life history of this species from 1983-1985 established that *T. bartschi* could grow and reproduce at ambient water temperatures in the Millstone area (report in preparation), but may require higher effluent temperatures for successful recruitment. The panels at 500 and 1000 m sampling sites were at depths of approximately 10 m and were not exposed to effluent water, under 2-unit operating conditions. No *T. bartschi* were found at 500 or 1000 m.

A similar link between high temperatures and the distribution of this species has been indicated by others; *Teredo bartschi* has been recorded outside its normal range only in the vicinities of thermal effluents. Hoagland and Turner (1980) reported that *T. bartschi* was present in panels at the mouth of the Waretown Creek and in Forked River from 1975 to 1978; both areas were in the mixing zone of the thermal plume from the Oyster Creek Nuclear Generating Station, Barnegat Bay, New Jersey.

In summary, a reproductive population of *Teredo bartschi* was collected at a site 100 m outside of the quarry cuts, within the mixing zone of 2-unit effluent. Preliminary data indicate that water temperatures above ambient are needed for larval recruitment, i.e., some incursion of effluent water; however, additional studies are needed to determine the exact conditions needed for *T. bartschi* settlement, and to determine whether the local distribution of *T. bartschi* will expand under 3-unit operating conditions. Additional studies may also substantiate the observed trend of increasing *T. navalis* abundance with decreasing distance from the quarry cuts.

TIMBER STUDY

MATERIALS AND METHODS

This study used five different types of wooden blocks or "timbers" which were exposed to the marine environment at five sites for periods of one to five years. The timbers (6.4 x 10.8 x 30.5 cm) were cut from planks (6.4 x 24 x 300 cm) commonly used for building docks. The five types of wood were: untreated Red Oak and Douglas Fir, and three types of chemically treated Southern Yellow Pine. The three chemical treatments were: 20 lbs per ft^3 creosote, 0.6 lbs per ft^3 chromated copper arsenate (CCA), and 2.5 lbs per ft^3 CCA. The timbers were deployed at four sites used for monitoring fouling and wood-boring species: White Point (WP), Fox Island (FI), Effluent (EF) and Giants Neck (GN), and at a fifth site (Fig. 1), located at the Niantic Bay Yacht Club, Black Point (NB). The timbers were fastened with plastic cable ties to wire lobster pots (Fig. 11). Three pots per site were deployed on the bottom with timbers arranged so that all untreated wood blocks were attached to the same pot.

Three timbers of each type were collected at each site in November and new ones deployed. Four replicate timbers of each type were placed at each site when this study began in March 1983, to be collected in November 1985, 1986, 1987, and 1988. These timbers, deployed for long-term exposures, were not replaced upon collection.

Wood-loss was quantified by comparing the weights of replicate timbers that have not been placed in seawater (blanks) with those that have been exposed. Every year, three blanks of each type of timber were cut from the same planks used to make timbers for deployment. After collection, the exposed timbers were processed according to the following procedure. First the percentage of surface area covered by each fouling species was recorded, and then the timbers were scraped clean and frozen. Next, they were radiographed using a 250 kV X-ray tube and the approximate percentage of wood lost was visually estimated from the radiographs. Wood-loss was expressed as the average of that percentage assigned to the top left, top right, bottom left and bottom right quadrants of each timber and was accomplished by rating the general proportions of bright areas, caused by various densities of shipworm tubes and the dark areas caused by various degrees of wood-loss. Finally, the timbers and the blanks were sectioned into one inch lengths and dried in a solar oven until they reached a constant weight. The untreated timbers were

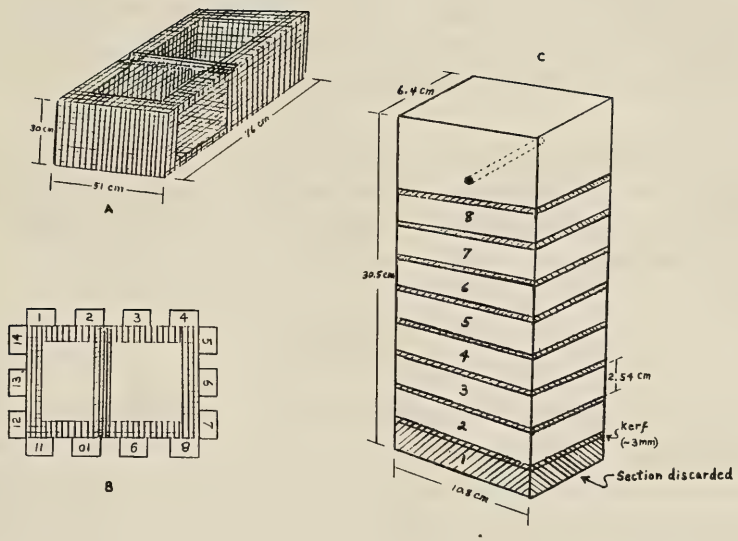

Figure 11. Diagram of a lobster pot (A), the location of timbers along its sides (B), and the dimensions of a timber with the location of sections which are weighed (C). These timbers are used to monitor wood-loss caused by marine woodborers in common building materials in the vicinity of the Millstone Nuclear Power Station.

acid-soaked and rinsed as described earlier for the exposure panels, but the chemically pre-treated timbers were not acid-soaked because of the resulting hazardous reaction products.

DATA ANALYSIS

The data summarize three sampling years, 1983-1985, prior to three-unit operation at MNPS. The first set of timbers was deployed in March 1983, and the last set was collected in November 1985.

The weight loss of exposed timbers was obtained by subtracting the weight of the exposed timbers from the average weight of 15 sections obtained from the three replicate "blank" timbers. In addition, since the dimensions of the timbers and the thicknesses of the sections varied from year to year, all weights were converted to a standard density unit, grams per cubic centimeter (g/cm^3).

In this report, the wood-losses for Douglas Fir and Red Oak were based on direct weighings. Direct weights of chemically treated timbers would have been biased by the calcium carbonate tubes of shipworms, as these timbers were not acid-soaked. Therefore, wood-losses for chemically treated woods were based on radiographic estimates.

RESULTS

There was considerable variability in the dimensions and quality of the planks used in this study. Of all the planks used, the Douglas Fir plank in 1983 and the creosote plank in 1984 were most dissimilar in their general dimensions. The quality of the pressure treated woods used in 1984 was poor. The creosote treated plank did not have 20 lbs of creosote per cubic foot because the density of these timbers (g/cm^3) was 39% lower than those in 1985. Similarly, the CCA treated sections in 1984 were dissimilar to those in 1985. Weight data are not available for unexposed pressure treated timbers in 1983. To account for all this variability, weight loss extimates were converted into grams per cubic centimeter (g/cm^3) of wood (Table 8).

The correlation between the wood-loss estimates by weight to that by visual inspection of radiographs was highly significant ($p > 0.001$). The second order polynomial equation used in the regression analysis has an R^2 of 0.81 (Fig. 12). The shape of this regression line relative to the data indicates that we consistently overestimated wood-loss using the radiographic method.

Wood-loss for Douglas Fir and Red Oak are presented in Figure 13. Douglas Fir lost an average of 3 times more wood during a one year exposure than did the Red Oak, ranging from 30-70% loss vs. 5-55% loss. In general, timbers that lost 70% of their weight were so fragile that several severely degraded timbers were lost from lobster pots.

Table 8. Average weight of a section for each type of "blank" timber used in the Timber Study at the Millstone Nuclear Power Station, from 1983-1985.

Type of wood	Year	Avg. section weight (g)	S E	C V	Avg. density (g/cm^3)	Average dimensions Length (cm)	Width (cm)	Height (cm)
Douglas Fir	1983	108.4	2.3	8.3%	0.43	11.9	8.1	2.6
	1984	86.8	0.9	4.0%	0.46	11.4	6.3	2.6
	1985	77.5	0.8	4.2%	0.43	11.1	6.2	2.6
Red Oak	1983	129.7	2.1	6.1%	0.62	11.7	6.9	2.6
	1984	112.2	0.9	3.2%	0.63	11.0	6.2	2.6
	1985	126.4	1.6	5.0%	0.75	10.8	6.0	2.6
Southern Yellow Pine								
0.6 lbs/ft^3 CCA	1984	112.8	0.9	3.1%	0.50	12.1	7.2	2.6
	1985	115.7	1.8	6.0%	0.64	11.1	6.3	2.6
2.5 lbs/ft^3 CCA	1984	111.7	1.0	3.6%	0.46	12.5	7.5	2.6
	1985	119.0	1.6	5.1%	0.67	11.0	6.2	2.6
20 lbs/ft^3 Creosote	1984	165.1	3.4	8.1%	0.58	12.1	9.4	2.5
	1985	169.5	3.1	7.1%	0.95	10.9	6.3	2.6

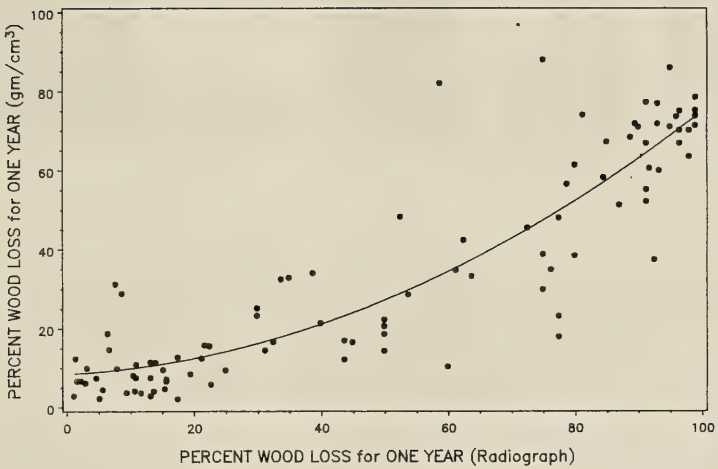

Figure 12. Comparison of wood-loss estimates from timber section weights versus visual assessments of radiographs for all types of wood examined from 1983-1985. These data were collected as part of the Exposure Panel Program at the Millstone Nuclear Power Station.

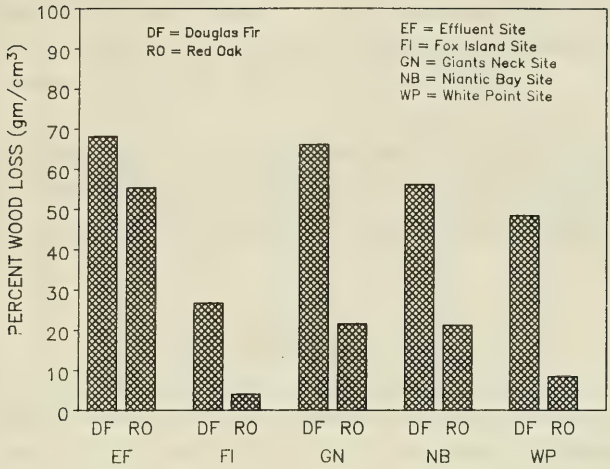

Figure 13. Mean annual wood-loss caused by marine woodborers from 1983-1985 in Douglas Fir and Red Oak timbers in the vicinity of the Millstone Nuclear Power Station. Estimates are based on the weight of five, one inch sections from each timber.

Radiographic estimates were used to describe wood-loss from chemically treated timbers. These data indicated that wood-loss was greatly reduced in treated woods relative to untreated types (Fig. 14). Only the cut surfaces of these timbers, which exposed areas with no chemical treatment, were susceptible to recruitment of woodborers. However, shipworms were able to penetrate some areas of CCA treated wood after they had settled and metamorphosed in untreated grains exposed by the cut surfaces.

Data concerning wood-loss from timbers exposed for more than one year indicated that wood-loss can more than double during the second year of exposure (Table 9). Generally, Douglas Fir timbers were totally degraded by woodborers after two years, while three years were required for Red Oak timbers to reach 70% wood-loss. At EF, wood-loss of untreated timbers was considerably greater, and neither

Douglas Fir or Red Oak timbers lasted through two years of exposure. At EF and GN, CCA treated timbers with untreated grain had up to 50% wood-loss after three years of exposure.

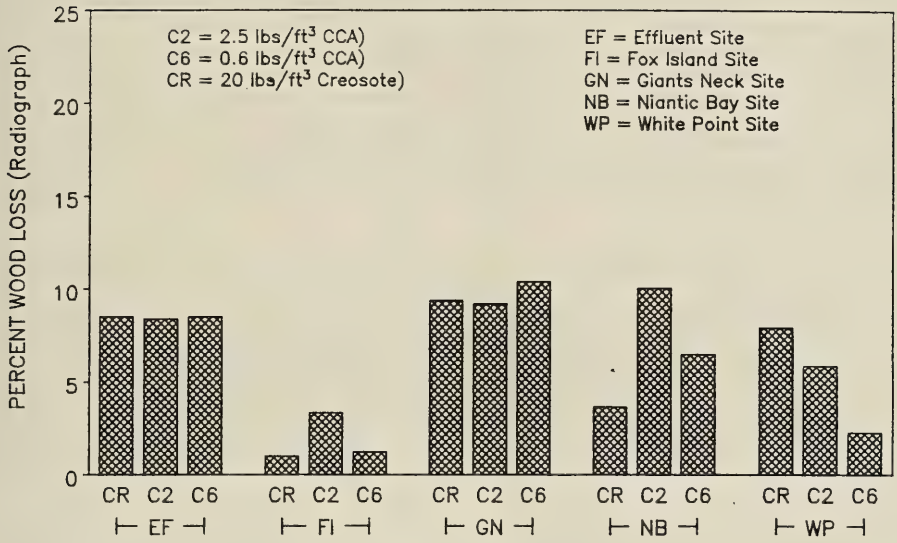

Figure 14. Mean annual wood-loss caused by marine woodborers from 1983-1985 in chemically treated Southern Yellow Pine timbers in the vicinity of the Millstone Nuclear Power Station. Estimates are based on visual assessments of radiographs.

Table 9. Percentage of wood-loss from timbers submerged for periods of 1, 2, and 3 years in the vicinity of the Millstone Nuclear Power Station, from 1983-1985.

Type of wood	no. of years exposed	Percentage of wood lost E F	F I	G N	W P
Douglas Fir	1	68%	27%	66%	48%
	2	*	82%	*	*
	1	55%	4%	21%	8%
Red Oak	2	*	49%	68%	56%
	3	*	70%	*	72%
Southern Yellow Pine					
0.6 lbs/ft^3 CCA	1	8%	1%	10%	2%
	2	12%	2%	---	2%
	3	44%	15%	---	---
2.5 lbs/ft^3 CCA	1	19%	1%	11%	10%
	2	28%	6%	19%	21%
	3	47%	26%	50%	---
20 lbs/ft^3 Creosote	1	0%	0%	0%	---
	2	0%	0%	0%	---
	3	---	---	---	---

NOTE: The Douglas Fir and Red Oak data are based on direct weights and the pressure-treated wood data are based on visual estimates of radiographs.
* Timbers were lost from pots and assumed totally destroyed by woodborers.
--- Timbers were lost from pots, but this loss was not related to woodborers.

DISCUSSION

The timber study quantified the annual loss of wood at the five sites in the vicinity of MNPS, and determined the longevity of five types of wood commonly used in marine construction. The untreated woods have provided direct weight data to compare wood-loss between sites, and the percentage of wood lost in the chemically treated woods has been calculated on the basis of visual estimates of wood-loss using radiographs.

The chemical treatments applied to the timbers protected the wood from woodborer attacks. However, woodborers readily recruited into the untreated wood grain exposed by the preparation of timbers. Within the first year, between 1-10% of the wood in the CCA treated timbers was lost and after three years

15-50% of the wood was lost. In addition, the adult shipworms appeared to be able to survive boring through the lower concentrations of CCA, which existed beneath the treated surfaces of the timbers. Woodborers were never observed penetrating the treated outside surfaces of a timber. The total protection of creosote timber from wood-loss in 1983 and 1985, resulted from the effective penetration of this chemical (20 lbs/cm^3) throughout the interior wood grain of each timber. Other studies have concluded that the CCA and creosote treated woods are very effective at deterring marine woodborers (Baechler et al. 1970; Johnson 1977; Richards 1977, 1979; Johnson and Gutzmer 1981). Creosote (20 lbs/ft^3) and CCA (1.0 lbs/ft^3) have been reported to withstand woodborer attacks for eight years, while test panels and piles with very high retentions of CCA (2.5 lbs/ft^3) have been reported to repel woodborers for 25 years (letter from W.T. Henry to J.D. Land, Koppers-Hickson Canada, Ltd., 1976). In their studies small test blocks were used and treated after being cut. In the present study, we used products and treatments available to local builders.

Annually, there was a threefold greater loss of wood from Douglas Fir timbers than from Red Oak timbers. Although Douglas Fir is seldom used in marine construction, oak planking and pilings are often used locally. The reduced wood-loss as observed for Red Oak is the reason hardwoods have been used in dock building. Oak pilings are much larger than our timbers and are set with the bark still on the tree, which provides additional protection from wood-borers. However, untreated woods in the local marine environments are readily attacked by borers.

Wood-loss varied from site to site, and FI had the least wood-loss of any of the sites sampled. However, Red Oak timbers at EF lost over twice as much wood when compared to the Red Oak timbers at ambient water sites. This was in contrast to data collected at EF using exposure panels, where wood-loss of panels at EF never exceeded that observed at ambient water sites. The reason for this discrepency was related to the location of the panels versus that of the timbers. Timbers were deployed directly on the bottom, while the EF panels were suspended off the bottom. Even the lower frame and rack assembly of panels deployed at EF was approximately one meter higher in the water column than were the timbers. The first inch of the timbers was usually embedded in the bottom sediments, which was the reason the first section of the timbers was routinely discarded during processing. Therefore, the greater annual wood-loss in timbers was caused by shipworms setting most heavily at the mudline, an occurrence documented by others (Grave 1928; Scheltema and Truitt 1956; Turner 1966; Nair and Saraswathy 1971).

In conclusion, our objectives for the timber study are being met. The untreated timbers have provided the needed database for quantifying wood-loss from site to site and year to year. The longevity of untreated Douglas Fir and Red Oak have been determined, and that for the treated woods continues to be investigated. Chemical treatments deter woodborer attack, but cutting treated timbers exposed unprotected surfaces. Of our test treatments, only creosote was penetrating enough to provide protection to interior wood grain. These data will be used to define potential power plant impacts on wood-loss in the vicinity of Millstone during three unit operation.

SUMMARY

1. The most abundant fouling species at ambient water sites (WP, FI, BP, GN) were *Balanus crenatus*, *Codium fragile*, *Cryptosula pallasiana*, *Botryllus schlosseri*, *Laminaria saccharina* and *Balanus eburneus* while the most abundant fouling species at EF (effluent site) were *Balanus improvisus* and *Mytilus edulis*. *Balanus eburneus*, a warm water barnacle, was consistently more abundant at EF than at the ambient water sites, and *L. saccharina* was never collected at the EF site. The identity and abundance of these fouling species did not affect abundance of woodborers.

2. Characteristics of the EF community which were related to temperature included enhanced primary cover, temporal shifts in peak abundance of individual species and total primary cover, absence of cold water species, and the unique occurrence of a warm water shipworm, *Teredo bartschi*.

3. *Teredo bartschi* recruited into EF panels after August with their largest populations occurring in panels collected in February. There is evidence to suggest that this species has a minimum setting temperature of approximately 22 °C.

4. *Teredo navalis* was most abundant at GN in May-Nov, while *Limnoria* spp. and *Chelura terebrans* were most abundant at WP in May-Nov. The low abundance of *T. navalis* at EF from 1979-1986 was caused by panel location. A second rack and frame assembly of panels will be relocated 0.2 m from the bottom in shallow water to further evaluate the vertical distribution of woodborers at EF.

5. The occurrence of *T. bartschi* at 100, 500 and 1000 m from the MNPS effluent discharge point was monitored. A reproductive population of *T. bartschi* was collected from panels at 100 m, after six months exposure.

6. Our data describe a consistent decrease in panel recruitment of *T. navalis* from 601 at 100 m to 388 at 1000 m. This trend may indicate that recruitment of *T. navalis* is enhanced in the effluent mixing zone.

7. Annually there was a threefold greater wood-loss from untreated Douglas Fir than from Red Oak timbers, but both types of wood were readily infested, and generally decomposed within 2-3 years.

8. Commercially available preservatives (e.g., creosote, CCA) retard woodborer infestation. However, chemical treatments were compromised by cutting the timbers. These cut surfaces exposed untreated grain which allow woodborers to enter the timbers. At EF and GN, CCA treated timbers with exposed untreated grain had up to 50% wood-loss in three years.

CONCLUSIONS

Results from these studies indicated that thermal effects due to two unit operation at the Millstone Nuclear Power Station were restricted to within 100 m of the effluent discharge point. In future studies, sustained increase in abundance of *Teredo navalis*, *Balanus eburneus* and *Mytilus edulis* or the sustained decrease in abundance of *Balanus crenatus* and *Laminaria saccharina* at nearby sites would indicate thermal effects caused by three unit operation. In addition, *Teredo bartschi* recruitment, and establishment of reproductive populations at sites beyond 100 m from the effluent discharge point would also be considered power plant related.

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Contents

FISH ECOLOGY 1
INTRODUCTION 1
MATERIALS AND METHODS 4
Ichthyoplankton program 4
Impingement program 8
Trawl program 10
Seine program 12
Data handling 16
Time-series analysis 18
RESULTS AND DISCUSSION 20
Ichthyoplankton 21
Impingement 26
Trawls 27
Seines 28
Selected species 30
Ammodytes americanus, American sand lance 30
Anchoa spp., anchovies 32
Gasterosteus spp., sticklebacks 36
Menidia spp., silversides 38
Microgadus tomcod, Atlantic tomcod 39
Myoxocephalus aenaeus, grubby 44
Tautoga onitis, tautog 46
Tautogolabrus adspersus, cunner 51
CONCLUSIONS 56
SUMMARY 57
REFERENCES CITED 58
APPENDIX 68

FISH ECOLOGY

INTRODUCTION

Finfish are an important marine resource and are found in a variety of habitats in the vicinity of Millstone Nuclear Power Station (MNPS) in eastern Long Island Sound (LIS). The construction and operation of MNPS could affect fish assemblages inhabiting the site environs (Fig. 1) by increasing mortality rates of various life stages and by altering spatial distributions. Populations may experience higher than normal mortality rates due to either impingement of young and adult fish on the intake screens, or entrainment of larvae in the cooling water system. The effect of increased mortality rates on these populations can be very different depending on the size, age structure and life span of the affected populations, and on the existence of compensatory mechanisms. Further, the spatial distribution of local fish populations may change in response to alterations in the thermal or chemical regime of the effluent or modifications to the physical habitat. Warmer water temperature can attract or exclude fish from areas affected by the thermal plume. Physical alterations caused by construction, dredging or bottom scouring could also attract or exclude fish from affected habitats. Because of these potential effects, Northeast Utilities Service Company (NUSCo) established several finfish sampling programs to provide baseline data for assessing the impacts of construction and operation of MNPS on local fish populations.

The objectives of the fish ecology programs are as follows:

1. To sample, identify, and enumerate fish found in the Millstone Point area;

2. To determine which fish species may be susceptible to impact from entrainment, impingement, or exposure to the heated effluent; and

3. To describe the fluctuations in abundance of life history stages of species that are potentially impacted and evaluate whether these fluctuations are within the expected historical range or have been effected by power plant operation.

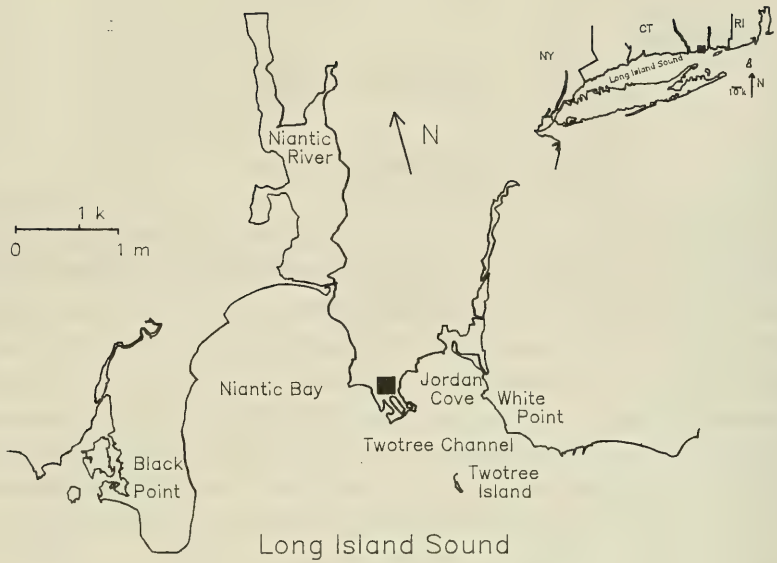


Figure 1. Location of the Millstone Nuclear Power Station in eastern Long Island Sound.

To meet these objectives the available life history stages of local finfish species are studied by various programs at Northeast Utilities Environmental Laboratory (NUEL). The sampling programs established over the years to monitor planktonic, demersal, pelagic and shore-zone fish abundances, were complemented by entrainment and impingement monitoring programs. Studies of planktonic fish eggs and larvae (i.e., ichthyoplankton) have been conducted at Millstone since 1973. These studies have included entrainment and offshore collections at various stations, and entrainment mortality and thermal tolerance studies on selected larval fish species. A trawl sampling program was established in April 1973 to monitor demersal fish. Since then, up to 11 stations have been sampled biweekly. Although the program has been reduced to six representative sites, the methodology continues today relatively unchanged. The seine program, established in 1969 to monitor shore-zone fish, involved sampling at up to 7 stations at various frequencies. This program has been reduced to three representative sites, but the methodology remains unchanged. Routine impingement sampling began at MNPS Unit 1 in 1972, and at Unit 2 in September 1975.

Monitoring at Unit 1 ceased with the installation of a fish return system in 1983. A gill-net program was established in 1971 to monitor pelagic fishes that were not caught in the trawl or seine programs, but it was discontinued in 1982 after an evaluation concluded that the gill-net program did not provide quatitative data, was not cost effective, and did not sample potentially impacted fish (see Appendices Ia and Ib and NUSCo 1982c).

The monitoring studies were supplemented over the years by several entrainment survival and fish diversion studies, and by an evaluation of the Unit 1 fish return (sluiceway) system. Special studies of mortality experienced by fish larvae during entrainment and the thermal tolerance of selected species were conducted (Carpenter 1975; NUSCo 1975). Mortality of larvae entrained through Unit 1 was estimated to range from 20 to 50%. Larvae were captured at the intake, discharge and quarry cut and held for 24 hr at intake water temperatures. Laboratory thermal tolerance studies, conducted on larvae of the silverside (*Menidia* spp.) mummichog (*Fundulus* spp.) and winter flounder (*Pseudopleuronectes americanus*) to assess thermal effects of power plant entrainment, indicated that mortality was low for silversides and mummichogs. The application of several devices (electric screens, noise generator, underwater lights, surface and bottom barriers and barrier nets) was investigated in an attempt to divert fish from the intakes and reduce impingement. Of the different devices tested, none reduced impingement (NUSCo 1976b). In 1980, NUSCo (1981b) demonstrated that it was practical and cost-effective to backfit a fish return system at Unit 1. A sluiceway was fabricated and installed there in December 1983, and results from a subsequent study indicated that survival of demersal fish and non-molting crustaceans exceeded 70% (NUSCo 1986b). Because the Unit 1 sluiceway has worked as designed and has successfully returned most organisms to LIS, it has mitigated the impact of impingement.

This report summarizes the monitoring data gathered by the ichthyoplankton, trawl, seine, and impingement programs during the period of two-unit operation, 1976 through 1985. Accounts of the evolution of these programs are also provided. In addition, the monitoring data and life history of eight potentially impacted fish species are presented and evaluated to determine if two-unit operation of MNPS has had any detrimental effect on them. Finally, it is noted that the monitoring data summarized in this report consitute the baseline against which three-unit operational data will be compared after 1986.

MATERIALS AND METHODS

Ichthyoplankton program

Ichthyoplankton studies have been conducted at Millstone since 1973. The program consisted of both "offshore" and "entrainment" sampling, and the number of samples collected by year and station is summarized in Appendix II.

Offshore sampling was initiated in 1973 to provide information for development and interpretation of entrainment impact predictive models (Sissenwine et al. 1973). A bongo sampler with two conical plankton nets of 0.333- and 0.505-mm mesh and weighted with a depressor was towed for 15 min. Various combinations of 16 stations (Fig. 2), sampling frequencies (weekly, biweekly, monthly), tow types (surface, sawtooth oblique, bottom), and times (day, night) were used (Battelle 1976) (Table 1). Following an evaluation of the program in 1975 (Vaughan et al. 1976), offshore sampling was redirected towards determining the densities and seasonal succession of the plankton community. The number of stations was reduced to six, the sampling frequency was changed to monthly, and night sampling was eliminated. The program was evaluated again in 1978 and the resulting recommendations were implemented in 1979. Sampling was limited to NB because this station provided the most representative samples of the offshore plankton community (NUSCo 1978). The 0.505-mm mesh net on one of the bongos was replaced with a 0.333-mm mesh net; this arrangement provided replication for the latter net. A stepped oblique tow (5 min each at surface, bottom, and mid-depth) provided a sample representative of the entire water column. Night sampling was reinstated to investigate the response of ichthyoplankton abundance to the diel period. The program has essentially remained unchanged since 1980, except for the use of a wire angle-indicator to more accurately position the net at the mid and bottom water depths.

The ichthyoplankton entrainment studies began in 1973 (Table 2). The density of fish eggs and larvae was estimated from three replicate samples collected at the intake (IN) at three depths: surface (in previous reports called INT1), mid (previously INT3) and bottom (previously INT5), at the discharge (EN, previously DIS1), and at the quarry cut (QCUT) (Fig. 3). Thirty plankton samples were taken every week (15 day, 15 night) with a 1.0 x 3.6-m conical plankton net with 0.333-mm mesh netting; the volume

Figure 2. Location of the offshore plankton sampling stations.

of water sampled was estimated from a TSK flowmeter. During 1974, studies were conducted to compare the catches from the 0.333-mm mesh offshore sampling net and the net used at EN, and to evaluate laboratory processing techniques. Results indicated that there was no significant difference between the abilities of the two nets to catch eggs or larvae nor in the techniques used to process the samples (NUSCo 1983). Beginning in July 1975, an electronic flowmeter was used to measure volume of entrainment samples. Sampling was limited to the discharge station, where flow velocities were high enough to eliminate problems of poor flowmeter response and net avoidance. Although the exact location of this station was somewhat different from that used previously, a comparability study revealed that there were no significant differences among density estimates made from samples collected simultaneously at the two sampling locations at the discharge (NUSCo 1976a). The sampling frequency was changed from one-day and one-night set of samples per week to three days and three nights per week (three replicates each, for a

Table 1. Summary of offshore plankton sampling program.

	J	F	M	A	M	J	J	A	S	O	N	D
1973					Stations 1-4, 6-10 & NB sampled [a] Weekly (day) and Monthly (night)							
1974	As in Dec. 1973	Stations 1-4, 6-13 & NB sampled[a]								Stations 1-4, 6-16 & NB sampled[a]		
	Weekly (day) and Monthly (night)											
1975	Stations 1-4, 6-16 & NB sampled [b]											
	Weekly (day) Monthly (night)					Biweekly (day) Monthly (night)				Monthly (day and night)		
1976	As in Dec. 1975	Stations 2, 6, 8, 11, 14 & NB sampled[b]										
1977 to 1978	Monthly (day only)											
1979 to Present	Station NB sampled [c]											
	Biweekly (day and night)				At least weekly (day and night)					Biweekly (day and night)		

[a] Sample technique – one 15-min sawtooth oblique towusing a bongo frame rigged with 0.333-mm mesh and 0.505-mm mesh plankton nets. Station NB previously reported as station 5.

[b] Same as "a", but replicate stratified (surface and bottom) and oblique tows were also done on a regular basis at randomly selected stations (see Battelle 1976 for details).

[c] In 1979 towing methodology consisted of 15-min sawtooth oblique tows using a bongo frame rigged with two 0.333-mm mesh plankton nets. Beginning in 1980 the methodology changed to a stepped oblique tow (5 min top, 5 min mid-depth, 5 min at bottom). From 1983 to present a wire angle-indicator was used to accurately position the net at surface, mid and bottom.

total of 18 samples per week) to improve the accuracy of density estimates and increase the sensitivity of subsequent analyses. When Unit 2 began using cooling water (fall 1976), sampling at EN began alternating weekly between the discharge structures of Units 1 and 2, when operating conditions permitted. The electronic flowmeter was replaced with an array of four General Oceanics flowmeters in 1980. The bases for replacing the electronic flowmeter were high cost, poor maintenance record, and inability to account for vertical and horizontal differences in flow observed at the discharges of the two Units (NUSCo 1983). The program was reviewed several more times (NUSCo 1981a, 1983, 1984a) and the sampling frequency reduced according to the schedule in Table 2. The evaluation of these sampling schedules (NUSCo 1983) indicated that no appreciable loss of accuracy had occurred despite a reduction in effort (see Appendix II).

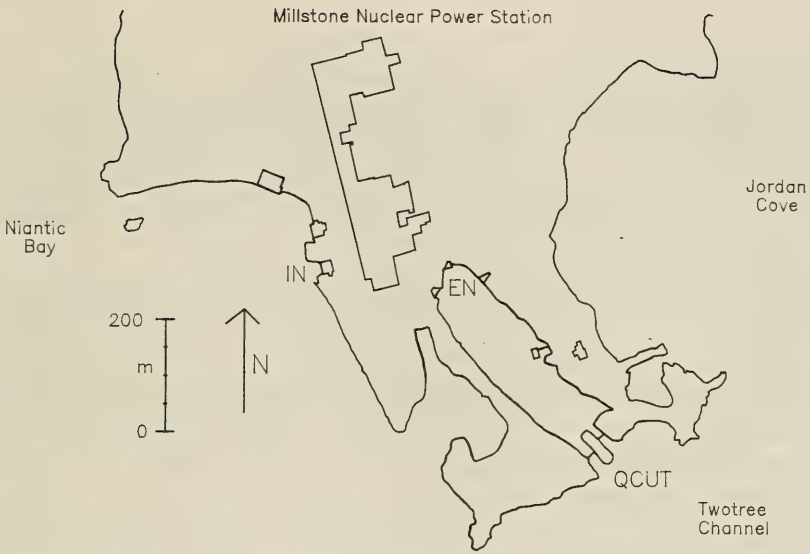


Figure 3. Location of the entrainment sampling sites.

The current ichthyoplankton sampling program includes weekly or biweekly collections at station NB (located in mid-Niantic Bay) and at least weekly collections at station EN. Samples at NB were taken with the bongo system described previously using 0.333-mm mesh nets. Sampling duration was 5 min at each depth (surface, mid, and bottom) in stepwise oblique tows. Sample volumes were estimated using one General Oceanics flowmeter in each net. Approximately 300 m^3 of seawater were filtered in each NB sample. Samples at EN were collected with the 1.0 x 3.6-m, 0.333-mm mesh conical plankton net deployed for 4 to 10 min on a gantry system. The deployment time depended on plant operating conditions and it was adjusted to filter about 400 m^3 of cooling water per sample. Four General Oceanics flowmeters were positioned in the mouth of the net to record flow. Volume sampled was calculated by averaging the volume estimates provided by the four flowmeters. All offshore and entrainment samples were preserved in a 5 to 10% formalin solution.

Although there have been minor changes in laboratory techniques to improve efficiency, the basic process has remained consistent. A dissecting microscope was set at 10 or 12 X magnification to view a portion of the sample and fish eggs and larvae were counted or removed using forceps. Initially, all fish

Table 2. Summary of entrainment plankton sampling program, 1973 - 1985.

	J	F	M	A	M	J	J	A	S	O	N	D
1973					3 reps[a] taken one day and one night per week at IN(surface, mid & bottom), EN & QCUT							
1974	3 reps taken one day and one night per week at IN(surface, mid & bottom), EN & QCUT											
1975	3 reps taken one day and one night per week at IN(surface, mid & bottom), EN & QCUT						3 reps[b] taken 3 days and 3 nights per week at EN					
1976 to 1980	3 reps[c] taken 3 days and 3 nights per week at EN											
1981 to 1982	3 reps taken 3 days and 3 nights per week at EN									3 reps taken 1 day and 1 night per week at EN		
1983 to 1984	1 rep taken 4 days and 4 nights per week at EN									1 rep taken 1 day and 1 night per week at EN		
1985	1 rep taken 1 day and 1 night per week at EN		1 rep taken 4 days and 4 nights per week at EN			1 rep taken 3 days and 3 nights per week at EN				1 rep taken 1 day and 1 night per week at EN		

[a] Volume estimated from TSK flowmeter readings

[b] Volume estimated from electronic flowmeter

[c] Volume estimated from General Oceanics flowmeter

larvae were removed from whole samples and identified to lowest practical taxon. Fish eggs were identified year-round beginning in May of 1979 and from only April through September from 1981 through the present. As the technology became available, a splitter (NOAA-Bourne, described by Botelho and Donnely 1978) was developed that allowed accurate subsampling and reduced sorting time. Successive subsamples were processed until at least 50 larvae and 50 eggs (for samples processed for eggs) were found, or until one-half of the sample was examined. Samples collected at EN were sorted for both fish eggs and larvae, but NB samples were sorted for larvae only (see Appendices III and IV). This further reduced laboratory processing time, but resulted in only minimal loss of information (NUSCo 1983). Cunner (*Tautogolabrus adspersus*) and tautog (*Tautoga onitis*) eggs were differentiated weekly using the criterion of bimodality of egg diameters (Williams 1967). Ichthyoplankton density was expressed as numbers per 500 m^3.

Impingement program

Fish impinged on the 9.5-mm mesh intake screens at Units 1 and 2 were periodically washed into perforated collection baskets. Screens wash at various intervals, depending upon debris loading, and

frequency ranges from continuous washing during storms to at least once every 8 h. Impingement samples were taken by sorting fish from all the material washed from the screens during a 24-h period, usually beginning and ending at about 0800. Fish were identified to the lowest possible taxon, counted, and up to 50 specimens of each species were measured to the nearest mm in total length. Catch was recorded as number impinged per 24-h period.

Routine impingement sampling began at Unit 1 in 1972, although some qualitative observations were made as early as 1971. Sampling at Unit 2 was initiated in September 1975. The primary objective of impingement monitoring at Millstone has been to quantify total annual species-specific mortality. Ways of minimizing this mortality were evaluated and plant design changes were recommended when appropiate. Throughout the 13 yr of monitoring, various changes have been implemented. These occurred mainly in four major areas, including the frequency of daily counts, the way in which fish lengths were recorded, the method used to estimate the number of fish impinged per 24-h period, and the elimination of impingement monitoring at Unit 1 after 16 December 1983 when a fish return sluiceway was installed.

Changes in the frequency of impingement monitoring are outlined in Table 3. From 1972 to March 1977, impinged organisms accumulated over a 24-h period and were counted daily. In 1977, sampling effort was reduced to 3 counts/week. Before this reduction, mean daily impingement estimates for each month based on 7 counts/week were compared to mean daily estimates extrapolated from 3 counts/week. The differences between actual monthly totals based on a complete census and the estimated totals ranged from 20 to 50%, depending upon the species (NUSCo 1978). At the level of effort reduced to 3 counts/week, more than 85% of all species were represented. In 1982, the impingement program was evaluated to determine if the precision of the impingement data could be improved by redistributing and optimizing effort (NUSCo 1983). Historical data (3 counts/week) were stratified by month and effort and reallocated according to El-Shamy (1979). The historical program (uniform effort - 3 counts/week) had a precision value of 0.79. When the sampling effort was hypothetically redistributed so that more samples were collected in those months when the variances of winter flounder (*Pseudopleuronectes americanus*) counts were high and fewer samples in months when variances were low, the precision factor increased to 0.88. An optimal sampling scheme was implemented in December 1983 (Table 3). Since then, sampling effort at Unit 2 was stratified by month so that 8 samples were collected in January, 15 in February, 14 in March, 5 in April, 4 per month from May through November, and 10 in December. The overall sampling effort was reduced by approximately 40%.

Table 3. Summary of impingement collections made per week. Unit 1 impingement collections were made from 1971, when it first became operational, through 1983. Impingement collections at Unit 2 were made from the time it became operational (1976) to the present.

	J	F	M	A	M	J	J	A	S	O	N	D
1971 to 1972	Irregular sampling											
1973 to 1976	7 24-h samples collected per week.											
1977 to 1983	3 24-h samples collected per week.											
1984 to date	2	4	3	1[a]	1	1	1	1	1	1	1	2[b]

[a] If necessary, additional samples were taken to ensure that 5 samples were collected during April.

[b] If necessary, additional samples were taken to ensure that 10 samples were collected during December.

During the first 3 yr of impingement monitoring (1972-1974), lengths of all fish impinged were recorded. From January 1973 to April 1975, length information was recorded for each species by categories: less than 3 in, 3 to 6 in, and greater than 6 in. This method did not accurately describe the sizes of various species and was discontinued. In May 1975, the present practice of measuring up to 50 individuals per sample was instituted. However, in 1977 and 1978, 100 or more individual lengths per species were recorded. Fifty measurements per sampling date for each of five taxa (winter flounder, silversides, threespine stickleback, grubby, and long-finned squid) were randomly selected from the data base. The resulting length frequency distributions were compared to the entire distributions for each species using chi-square goodness-of-fit tests. For all five species, the random length distributions with 50 measurements were not significantly different ($p > 0.98$) from the original distribution (NUSCo 1983).

Trawl program

The trawl sampling program was established in April 1973 to monitor demersal fish. Throughout the program, demersal fish were sampled using a 9.1-m otter trawl with a 0.6-cm codend liner. Initially, one 15-min haul was made biweekly at each of seven stations, including 1, 4, JC (previously called station 6), 7, TT (previously station 8), 9 and 10 (Fig. 4). In September 1974, station IN (previously station 11) was added because it was a potential location for the Unit 3 intake. In February 1975, station NB (previously station 5) was added to provide a more representative sample from Niantic Bay. Sampling began at station NR (previously station 2, then N2) in June of 1975 to obtain information on fish communities in the lower Niantic River. In February 1976, station BR (previously station 14) was added

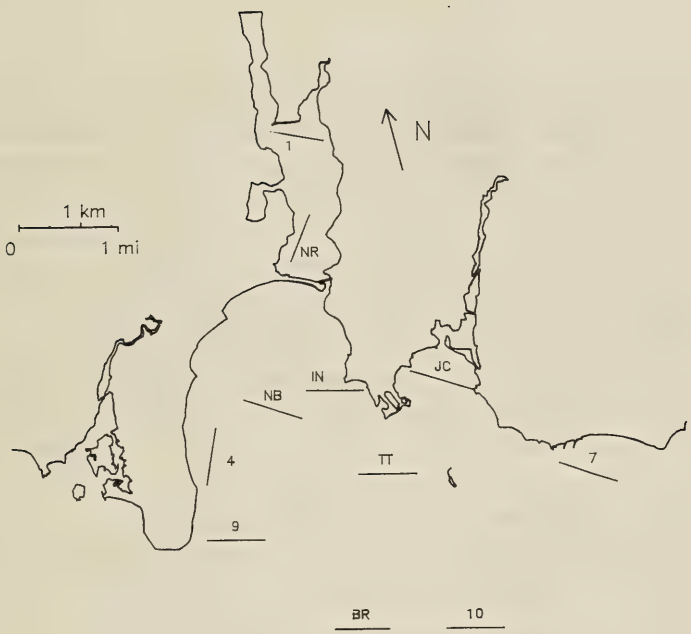


Figure 4. Location of the trawling sampling stations.

because the area was considered another potential site for the Unit 3 intake. Single and duplicate tows were made at various sites from 1973 to 1976 and are outlined in Table 4 along with a summary of the addition and deletion of sampling stations. Several stations were eliminated in 1976: stations 1, 7, and 10 because the bottom cover made trawling inefficient; station 4 because it was similar to NB; and station 9 because it was very close to BR. Since February 1976, triplicate tows have been made biweekly at NR (Niantic River), NB (Niantic Bay), JC (Jordan Cove), TT (Twotree), IN (Intake), and BR (Bartlett Reef). These six stations are believed to represent the different demersal environments surrounding Millstone Point.

In October 1977, the unit of trawl effort was changed from 15 min to 0.69 km over the bottom because it was felt that demersal fish abundance would be better estimated using a unit of effort based on the area swept. The distance 0.69 km was used because it was the maximum distance that could be covered at JC. This distance also provided some continuity with previous data because it approximated

the length covered in 15 min when the boat was towing a trawl under average conditions, with the engine at idle speed and not influenced by tidal currents. Up to 50 individuals of a taxon at each station were measured (total length) to the nearest millimeter. Catch was expressed as number of fish per standardized tow.

Table 4. Summary of trawl sampling program. The station numbers correspond to the stations listed in Figure 4. All stations were sampled biweekly with a 9.1-m Wilcox otter trawl. Stations: 1 = Upper Niantic River; NR = Lower Niantic River; 4 = Crescent Beach; NB = Niantic Bay; JC = Jordan Cove; 7 = Seaside; TT = Twotree; 9 = Black Point; 10 = Outer Bartlett Reef; IN = Intake; BR = Bartlett Reef.

	J	F	M	A	M	J	J	A	S	O	N	D
1973			Stations 1,4,7,9,10,JC & TT --- One 15-min tow at each station.									
1974	Stations 1, 4, 7, 9, 10, JC & TT: one 15-min tow at each.					Two tows at one station.[a]		3 tows at one sta.[a]	Sta. 1, 4, 7, 9, 10, JC, TT & IN: Two tows at TT, 10, IN; One tow elsewhere.			
						One 15-min tow at other stations.						
1975	As in Dec. 1974	Stations 1, 4, 7, 9, 10, JC, TT, NB, IN				Stations 1, 4, 7, 9, 10, JC, NB, TT, IN & NR:						
		Two 15-min tows at Station 10, NB, TT, IN and one random station; one tow elsewhere.										
1976	As in Dec. 1975	Stations 1, 4, 7, 9, 10, JC, TT, NB, IN, NR, BR: Three 15-min tows at JC, NB, TT, IN, NR, BR; one 15-min tow elsewhere										
1977	Stations JC, TT, NB, IN, NR, BR: three tows at each station. (15-min)								(0.69 km over bottom)			
1978 to present	Stations JC, TT, NB, IN, NR, BR --- Three tows at each station (0.69 km over bottom)											

[a] This station was selected randomly.

Seine program

The seine sampling program, established in 1969 to monitor shore-zone fish, is summarized in Table 5. Shore-zone fishes were sampled using a 9.1 x 1.2-m knotless nylon seine net of 0.6-cm mesh hauled parallel to the beach for about 30 m; three replicates were taken in adjacent areas of the beach. Because a preliminary study (Battelle 1973) indicated that there was no difference in the numbers of fish collected

from the shore-zone at different tidal stages, all collections were made in the 2 h before high tide. The shore-zone fish assemblages of seven different areas were sampled at some point in time since May 1969. These areas were GN (Giants Neck), BL (Black Point), SP (Sandy Point in the Niantic River), IN (Bay Point in front of the station intakes), JC (Jordan Cove), WP (White Point), and SS (Seaside) (Fig. 5). From 1969 through 1972, stations GN, IN, JC, and WP were sampled. In February 1973, SS and BL were added. Station SP was sampled in only 1975. Sampling at IN was discontinued in 1983 when construction activities associated with the Unit 3 intake removed a major portion of the beach. Sampling at BL was also discontinued in 1983 because its bottom type did not correspond to other stations and because few fish were caught there. Catches of shore-zone fish at SS also did not correlate with catches at other stations, probably because the station was more exposed to wind and wave action than the others (NUSCo 1983). Sampling at SS was discontinued in 1984 and effort was redistributed to the special seine study described below. Initially, hauls were made in February, May, July, September, and December; this frequency continued through 1973. In 1974, sampling frequency was increased to include collections during June, August, and October. Collections in November were added in 1982 and in January during 1984. Also, in 1984 sample frequency was increased to biweekly from April through October.

Figure 5. Location of the shore-zone seine sampling stations.

Throughout the sampling program, fish were identified to lowest practical taxon and measured to the nearest millimeter in length. From 1969 to 1980, standard length was measured. During 1981-1982, both standard and total lengths were measured and a regression was used to convert previously recorded standard lengths to total lengths (NUSCo 1984a). When more than 50 individuals of a taxon were collected in a replicate, a representative subsample was measured; otherwise all fish were measured. Catch was expressed as number of fish per 30 m haul.

Special seine study. In 1982, the seine program was evaluated to determine if it would adequately address the potential impact of the three-unit thermal plume. The sampling frequency was increased to biweekly from April through October to achieve a 50% detectability level for the dominant shore-zone taxon, the silverside. MacCall et al. (1983) recommended this level as a criterion for long-term impact assessment programs. Because the thermal plume was projected to encompass Jordan Cove and raise the water temperature 0.3°C on the flood tide and 1.2°C on the ebb, sampling was scheduled for both tidal stages from April through October, when most fish are in the shore zone. Initially (April-July 1984),

Table 5. Summary of seine sampling program during 1969-1985. Three seine hauls were made with a 9.1-m, 0.63-cm mesh seine net at each station (see Figure 5). Unless otherwise indicated, all samples were collected during the 2-h period preceeding high tide.

	J	F	M	A	M	J	J	A	S	O	N	D
1969					GN WP IN JC		GN WP IN JC		GN WP IN JC			GN WP IN JC
1970 to 1972		GN WP IN JC			GN WP IN JC		GN WP IN JC		GN WP IN JC			GN WP IN JC
1973		GN WP IN JC SS BL			GN WP IN JC SS BL		GN WP IN JC SS BL		GN WP IN JC SS BL			GN WP IN JC SS BL
1974 to 1982[a]		GN WP IN JC SS BL			GN, WP, IN, JC, SS, BL Monthly							GN WP IN JC SS BL
1983		GN WP IN JC SS BL			GN, WP,JC, SS Monthly							
1984	GN, JC, WP, SS Monthly			[b]	GN, WP-- Biweekly, High + Ebb SS-- Monthly, High JC-- Biweekly High, Monthly Ebb						GN, WP, SS, JC Monthly	
1985	GN WP JC Monthly				GN WP -- Biweekly, High + Ebb JC -- Monthly, High + Ebb						GN WP JC Monthly	

[a] During 1975 only one haul was made at IN, three hauls were made at SP.

[b] During this time period, JC and WP were sampled biweekly, high + ebb; GN and SS were sampled monthly, high.

stations JC and WP were sampled biweekly on both tides. Both sites are within the area projected to be influenced by the thermal discharge from three-unit operation. But because only 2 yr of 3-unit pre-operational data could be collected on the ebb tide, a new sampling scheme was adopted based on a control-treatment pairing (CTP) design (Skalski and McKenzie 1982; Bernstein and Zalinski 1983). The critical requirement of the CTP design is the selection of the control-treatment (nonimpacted-impacted) pairs of stations where the abundance of an organism responds similarly to changes in environmental parameters. The ratios of control-to-treatment data from each pair of stations, are compared between pre-operational and operational phases to detect impacts. An analysis of 15 yr of data showed significant correlations between GN (control) and WP (treatment) data using silverside and total fish abundance as response variables. Based on the results of this analysis, a new sampling scheme was adopted. Our control-treatment pair (GN, WP) is now sampled biweekly. Because work to date will serve as a baseline for a future assessment of three-unit operation, no data from this special study are presented here.

Data handling

To assess impacts it was necessary to identify potentially affected species, document their spatial distribution, and describe the natural temporal fluctuations of their life history stages collected near MNPS. Although sampling at NUEL has occurred since 1969, the changes in each program have limited the comparability and usefulness of data collected before 1976. Therefore, when data were available, analyses were restricted to 1976-1985, the period of two-unit operation at MNPS. The only exception to this was that the 1969 through 1985 seine data were used in the time-series analysis of silversides.

The selection of potentially affected species was based on their prevalence in entrainment or impingement samples. Spatial distribution patterns were based on the catch of a species at each station. Temporal fluctuations were described by annual (median and/or mean) catches and by the forecasts provided by time-series models, which used log-transformed catch data. The distribution of life history stages in impingement, trawl, and seine collections was inferred from the morphological characteristics and sizes of the fish. Identification of each fish species in all programs was made to the lowest possible taxon. Some specimens were identified to genus or family if they were juveniles, adults that could not be easily identified in the field, or if they were species of uncertain taxonomic status due to inadequate descriptions in the literature. The taxa which included more than one species are listed in Table 6. This table also includes the programs in which each taxon was used, the reason for combining species, and the probable species that were included in each group identification.

Table 6. List of taxa which were identified to genus or family.

Taxa	Program[a]	Reason[b]	Possible identification
Alosa spp.	PL,I,T	1,2,3,4	*Alosa aestivalis, Alosa mediocris, Alosa pseudoharengus, Alosa sapidissima*
Anchoa spp.	PL,I,T,S	1,2,3	*Anchoa hepsetus, Anchoa mitchilli*
Bothidae	PL,T	3,5	*Bothus ocellatus*, Left-eyed flounder
Clupeidae	PL,I,T,S	1,4	*Brevoortia tyrannus, Clupea harengus, Alosa* spp.
Fundulus spp.	PL,I T,S	3	*Fundulus majalis, Fundulus heteroclitus*
Gadidae	PL,I,T,S	2,3	*Enchelyopus cimbrius, Gadus morhua, Microgadus tomcod, Pollachius virens*
Gasterosteidae	PL,I,T	3	*Apeltes quadracus, Gasterosteus aculeatus, Gasterosteus wheatlandi, Pungitius pungitius*
Gobiidae	PL,T	3	*Gobionellus boleosoma, Microgobius thalassinnus, Gobiosoma bosci, Gobiosoma ginsburgi*
Liparis spp.	PL,I,T	1,3	*Liparis atlanticus, Liparis liparis*
Menidia spp.	PL,I,T,S	1,2,3	*Menidia beryllina, Menidia menidia*
Myoxocephalus spp.	PL,I,T	2,3	*Myoxocephalus aenaeus, Myoxocephalus octodecemspinosus*
Prionotus spp.	PL,I,T,S	3	*Prionotus carolinus, Prionotus evolans*
Sciaenidae	PL,I,T	2,5	*Bairdiella chrysoura, Cynoscion regalis, Leiostomus xanthurus, Menticirrhus saxatilis*
Urophycis spp.	PL,I,T,S	1,2,3	*Urophycis chuss, Urophycis regia, Urophycis tenuis*
Raja spp.	I,T	1	*Raja erinacea, Raja ocellata*

[a] Program: PL = plankton (larval), I = impingement, T = trawls, S = seines

[b] Reason: 1 = Difficult to identify using external features; 2 = Juveniles similar; 3 = Incorrect identification may have occurred; 4 = literature is not clear at species level; 5 = Too few collected for exact identification.

Ichthyoplankton abundance estimates. Because the ichthyoplankton data collected had skewed distributions, median rather than arithmetic densities (no./500 m^3) of the most abundant icthyoplankton taxa entrained (station EN) were used to describe temporal trends. These medians, also used for calculating entrainment estimates, were determined from data collected during the period when each species occurred annually. The period of occurrence was that time during which 95% of the annual cumulative abundance occurred; the data falling in the two tails (2.5% each) of the cumulative frequency distribution were not used to compute the medians. Median densities could not be calculated for any taxon that had many zero observations within its annual period of occurrence because the resulting median would have been at or near zero. Therefore, arithmetic mean densities (no./500 m^3) were computed and used as an index of relative abundance to rank all the species in the species list tables. Similarly, annual mean densities were used to determine annual cumulative densities for the six most abundant taxa of ichthyoplankton.

Entrainment estimates. Annual entrainment estimates for selected species of larvae were calculated from 1976 through 1985 and for eggs from 1979 through 1985. These estimates were obtained by multiplying the median density at EN during the period when 95% of the annual cumulative abundance occurred times the total volume of water passed through MNPS during the same period. A nonparametric method (Snedecor and Cochran 1967) was used to construct a 95% confidence interval around each median density and corresponding entrainment estimate.

Impingement estimates. Historically, several methods were used to estimate impingement when counts were no longer made daily after April 1977. The first method used the real-time to sample-time ratio to obtain estimates. Beginning in 1979, monthly impingement estimates were based on the extrapolation of actual counts using a volumetric ratio. The daily cooling-water volume was calculated based on 15 min flow rates from 0000 to 2345 for the date on which the impingement sample was taken during the morning. This flow rate was then lagged back one day before it was used in the estimation procedure. The present estimation procedure was developed in 1985 to account for the fact that impingement rates are directly influenced by cooling water flow (Con Ed and PASNY 1977; Lawler, Matusky and Skelly Engineers 1980). Cooling-water volume estimates corresponding to the actual 24-h impingement period (0800 to 0745) of each sample were used in order to improve accuracy. Within each month, an estimate for every day not sampled was calculated by multiplying the average impingement density (number of fish per m^3 of cooling water) based on the days sampled in that month times the volume of cooling water on each day not sampled. All of these daily estimates were then added to the sum of the actual sample counts to arrive at the monthly totals for each species. Annual impingement estimates were calculated by summing the monthly estimates.

Fish length frequency data. As indicated previously, sampling effort was stratified by season for seine sampling and by month for impingement sampling during some periods of the study. Therefore, whenever appropriate, the length-frequency data were weighted to account for unequal effort during the year. Because seine sampling effort from April through October was twice that during the remainder of the year, data collected from November through March were weighted by a factor of two. For impingement collections, monthly weight factors of 4 (January), 2 (February and March, 6 (April), 7 (May through November), and 3 (December) were used to standardize monthly effort close to 30 (range of 28 to 32).

Time-series analysis

Autoregressive integrated moving average (ARIMA) time-series models were developed to describe the natural fluctuations of potentially impacted species in the MNPS area during the period spanned by our baseline data. Because the model building process and analytical aspects of this technique have been discussed in detail elsewhere (Bireley 1985, 1987; NUSCo 1985), only a brief review of the methodology follows.

ARIMA models for each species were fitted only to time-series data from those stations and life history stages where occurrence was high enough to provide reasonable descriptions of natural fluctuations. The data (densities from ichthyoplankton, catches from impingement and seines) were log transformed to reduce skewness and to stabilize variances (Glass et al. 1975), and then averaged over the sampling period specific to each program to ensure that data would be spaced at equal intervals in the final time-series. The sampling periods were a week for impingement and ichthyoplankton data and a month for seine data.

The deterministic portion of the ARIMA models included explanatory variables much like nonlinear regression models. These variables were cooling-water flow (F), species season indicator (S), and periodic components which entered the model as sine or cosine functions of 1 to 6, or 12, 24, 36 ... up to 120 months; or combinations of these periods. The stochastic portion of the model described the structure of the model prediction errors using two types of stochastic terms, autoregressive (A) and moving-average (M). Therefore, the general form of the ARIMA models was:

$$Z_t = Q_t\left[I + \sin(tK_p) + \cos(tK_p) + \ldots.\right] + \left[\text{Stochastic terms: } A_t + M_t + \ldots.\right]$$

where Z_t were the time-series of means of the log transformed data; the subscript (t) was time in days; the multiplier Q of the deterministic portion of the model could be either the flow (F) for impingement models or the season indicator (S) which had a value of zero when a species was known to be absent from the MNPS area and a value of one otherwise (Table 7); and (I) was a constant estimated from the data either as zero or as unity. The periodic component K_p was a constant that converted the time (t) into angular units (radians) of some specified period (p) in months. Some models had more than one pair of sine-cosine terms to accomodate more than one periodic component.

A stepwise regression procedure that maximized R^2 values was used to select the best combination of the above variables for the deterministic portion of each ARIMA model. Next, appropriate stochastic terms were added to the deterministic model whenever the residuals were found autocorrelated. The methods of Brocklebank and Dickey (1986) were used to identify the stochastic structure of the residuals from the deterministic function.

Interpretation of time-series model statistics. Because the time-series models presented in this report characterized the abundance levels and natural fluctuations of local fish populations during the two-unit operation period (1976-1985), the model forecasts consitute the two-unit standard- or reference-series against which future three-unit operational monitoring data will be compared. Several summary statistics reported with the results of the time-series models will be used for impact assessment purposes in the future. The "errors" or deviations of each data point from the model forecast were separated into "above" (positive deviations) and "below" (negative deviations) and added up by years. These sums of deviations indicated whether the year being examined was above or below the two-unit reference series. When the deviations are squared, the annual sums become the annual components of the model sum of squares error (SSE). The mean squared errors (MSE's) were obtained by dividing the SSE's by the number of degrees of freedom associated with each year and model. These MSE values provide a basis for impact assessment because any annual MSE can be statistically compared to the reference series model MSE by means of a simple F-test obtained as the ratio of the two MSE's. It is expected that the annual fluctuations of MSE during the three-unit operation period, will be within the range of MSE values reported for this preoperational period. Note that the periods modeled at EN and NB are not the same (1976-1985 versus 1979-1985). This may cause the summary statistics to show different patterns of deviations from the models that characterize temporal fluctuations at the two stations (Appendices XIV through XXVIII).

Table 7. The species and program combinations for which the multiplier variable for season (S) in their time-series models was set equal to one.

Species	Program	Season
Ammodytes americanus	Larval	November - July
Anchoa spp.	Larval	May - November
	Egg	April - August
Menidia spp.	Seine	May - December
Myoxocephalus aenaeus	Larval	January - June
Tautogolabrus adspersus	Larval	May - October
	Egg	April - August
Tautoga onitis	Larval	May - October
	Egg	April - August

RESULTS AND DISCUSSION

The fish studies at MNPS include data on over 100 taxa from icthyoplankton, impingement, trawl, and seine samples collected from January 1976 through December 1985 (Appendices V through XVII). The most common of these were: *Ammodytes americanus* (American sand lance), *Anchoa* spp. (anchovies), *Gasterosteus* spp. (sticklebacks), *Menidia* spp. (silversides), *Microgadus tomcod* (Atlantic tomcod), *Myoxocephalus aenaeus* (grubby), *Raja* spp. (skates), *Stenotomus chrysops* (scup), *Scophthalmus aquosus* (windowpane), *Tautoga onitis* (tautog) and *Tautogolabrus adspersus* (cunner). These taxa were typical of fish assemblages found in other areas of LIS (Baird 1873; Bean 1903; Greeley 1938; Warfel and Merriman 1944; Merriman and Warfel,1948; Wheatland 1956; Richards 1959, 1963; Richards et al. 1963; Pearcy and Richards 1962; Perlmutter 1971; McHugh 1972; Saila and Pratt 1973) and the northeast (Bigelow and Schroeder 1953; Briggs 1975; Fritzsche 1978; Leim and Scott 1966; McHugh 1977). Around the Shoreham Nuclear Power Station in mid LIS, Austin and Amish (1974) reported a similar composition of ichthyofauna, but they found that *Cynoscion regalis* was an abundant larval taxon. NAI (1979) and CT DEP (P. Howell, pers. comm) also reported that these taxa were abundant among LIS ichthyofauna.

Eight of the above taxa were selected for a detailed analysis based on their susceptibility to impact from impingement and entrainment. The selected taxa were those that contributed at least 3% to the total catches from either impingement or entrainment (Table 8 and Appendix V). Although winter flounder met this criterion, this species was not included in the above mentioned group of eight taxa because it is discussed in a separate report section (Winter Flounder Studies). American sand lance ranked first among impinged taxa and third among entrained larval taxa. Larval anchovies ranked first in

Table 8. Percentage contributed by each taxon to the estimates of total entrainment and impingement at MNPS during 1976-1985.

	Entrainment		
Taxa	Larvae	Eggs	Impingement
Anchoa spp.	61.3	10.8	8.1
Pseudopleuronectes americanus	10.8	< 0.1	8.5
Ammodytes americanus	9.7	< 0.1	47.8
Myoxocephalus aenaeus	3.7	< 0.1	5.9
Tautogolabrus adspersus	1.8	55.5	1.9
Tautoga onitis	1.8	27.2	0.8
Menidia spp.	0.1	0.1	5.5
Microgadus tomcod	0.1	0.0	3.4
Gasterosteus aculeatus	< 0.1	0.0	5.1
Gasterosteus wheatlandi	< 0.1	0.0	1.7

entrainment. Eggs of cunner and tautog were the two most abundant egg taxa entrained; cunner was also often caught in other sampling programs. Silversides ranked fifth in impingement. The grubby ranked fourth among both impinged and entrained larval taxa. Sticklebacks and Atlantic tomcod each contributed over 3% to the total impingement.

Results for the two-unit operational period (1976-1985) are summarized separately for the ichthyoplankton, impingement, trawl, and seine sampling programs, and are followed by a discussion for each of the eight selected taxa. Although the analytical techniques previously described were applied to the data of all eight taxa, not all of the analytical results provided useful interpretations for impact assessments. Therefore, the results and conclusions that follow may be based on different techniques depending on the sampling program and taxon.

Ichthyoplankton

Because natural ichthyoplankton mortality rates are one of the most important controlling factors of adult fish stock abundance (Cushing and Harris 1973; Bannister et al. 1974; Cushing 1974; May 1974; DeAngelis et al. 1977), additional mortality due to entrainment could affect local fish populations. Thus, the plankton studies conducted at Millstone since 1973 have included ichthyoplankton entrainment estimates, density indices, and species composition. The data summarized in this section are from the period 1976 through 1985 and restricted to stations EN and NB.

Annual entrainment estimates (based on medians) for the most common larval and egg taxa are presented in Table 9. Anchovies were the most abundant fish larva entrained; annual estimates ranged from a minimum of 1.5×10^6 in 1984 to a maximum of $1{,}284 \times 10^6$ in 1981, and the 10-yr total was $4{,}056 \times 10^6$ larvae. The numbers of larval sand lance and grubby entrained during 1976-1985, totaled 359 and 313×10^6 respectively and were an order of magnitude lower than the estimates for anchovies. During the past 7 yr, more cunner eggs were entrained than any other taxon ($13{,}480 \times 10^6$). Median densities were relatively constant and entrainment estimates ranged from 1,610 (1981) to 2,589 (1983) $\times 10^6$ eggs (Table 9). The total entrainment estimates of tautog and anchovy eggs were about one-half and one-fifth, respectively, of that for cunner.

Table 9. Annual median density (no./500m^3) at EN and estimated entrainment of selected taxa.

Ammodytes americanus larvae

Year	Median density	Lower 95% CI	Upper 95% CI	Num. days sampled	Entrainment estimate[a]	Lower 95% CI[a]	Upper 95% CI[a]
1976	16.2	12.8	20.4	103	18.6	14.7	23.5
1977	48.9	41.5	56.9	140	66.7	56.6	77.5
1978	58.2	38.0	77.8	116	36.4	21.9	48.6
1979	50.7	42.6	61.4	143	62.4	52.4	75.6
1980	46.6	40.3	58.7	137	66.6	57.6	83.8
1981	73.7	64.1	86.5	130	57.4	50.0	67.4
1982	11.4	9.6	14.4	113	12.4	10.4	15.6
1983	15.5	11.0	21.4	121	19.6	14.0	27.1
1984	10.1	8.1	12.2	132	13.1	10.4	15.7
1985	5.3	5.0	7.9	124	5.8	8.8	8.6
Total					359.0		

Anchoa spp. larvae

Year	Median density	Lower 95% CI	Upper 95% CI	Num. days sampled	Entrainment estimate[a]	Lower 95% CI[a]	Upper 95% CI[a]
1976	680.0	510.2	880.0	65	448.4	334.2	576.4
1977	215.4	157.7	329.3	71	162.5	119.0	248.4
1978	248.4	173.3	367.9	59	160.0	111.6	236.9
1979	1020.8	734.9	1302.0	60	600.9	432.6	766.5
1980	1044.6	900.2	1323.5	51	558.1	480.9	707.1
1981	2285.0	1888.9	2725.9	51	1284.1	1061.5	1531.9
1982	429.1	328.8	567.4	64	299.7	229.6	396.3
1983	801.7	571.8	1119.6	76	485.6	346.3	678.2
1984	137.6	89.3	240.2	61	1.5	0.9	2.7
1985	581.0	479.7	958.3	73	454.9	375.6	750.3
Total					4055.7		

Anchoa spp. eggs

Year	Median density	Lower 95% CI	Upper 95% CI	Num. days sampled	Entrainment estimate[a]	Lower 95% CI[a]	Upper 95% CI[a]
1979	850.6	670.8	989.6	55	464.1	366.0	540.0
1980	449.3	115.9	614.6	38	183.1	47.2	250.4
1981	666.1	514.4	833.5	50	369.3	285.2	462.1
1982	473.5	328.3	614.1	42	213.6	148.1	277.0
1983	1563.7	1081.4	2174.5	42	503.5	348.2	700.2
1984	2401.2	1154.4	3715.8	37	807.7	388.3	1249.9
1985	22.6	0.0	75.8	74	16.0	53.7	
Total					2557.3		

[a] x10^6 of fish larvae or eggs.

Table 9. (continued).

Myoxocephalus aenaeus larvae

Year	Median density	Lower 95% CI	Upper 95% CI	Num. days sampled	Entrainment estimate[a]	Lower 95% CI[a]	Upper 95% CI[a]
1976	17.9	15.0	22.5	73	12.0	10.1	15.2
1977	44.9	37.1	50.2	67	30.2	24.9	33.7
1978	18.4	15.2	22.3	90	8.9	7.3	10.8
1979	30.7	27.7	34.3	84	19.8	17.9	22.2
1980	30.8	26.3	37.9	90	30.2	25.8	37.1
1981	98.5	73.1	113.0	73	45.0	33.4	51.7
1982	59.2	53.3	73.0	78	46.4	41.8	57.2
1983	62.1	51.6	78.8	79	50.0	41.6	63.5
1984	45.2	36.6	53.9	82	35.8	29.0	42.7
1985	64.4	51.0	75.0	72	36.5	28.9	42.5
Total					312.9		

Tautoga onitis eggs

Year	Median density	Lower 95% CI	Upper 95% CI	Num. days sampled	Entrainment estimate[a]	Lower 95% CI[a]	Upper 95% CI[a]
1979	919.9	724.3	1153.4	80	645.8	508.5	809.7
1980	1608.4	1356.2	1277.7	72	992.1	836.5	1158.2
1981	1618.9	1404.1	1934.5	86	1385.3	1201.5	1655.4
1982	1562.9	1278.9	1709.7	84	1443.4	1181.1	1579.0
1983	1309.0	986.4	1750.2	92	953.7	718.7	1275.2
1984	1486.4	1123.4	1892.8	88	1211.9	915.9	1543.2
1985	1519.9	1133.1	2844.8	92	1260.1	939.4	2358.6
Total					7892.3		

Tautogolabrus adspersus eggs

Year	Median density	Lower 95% CI	Upper 95% CI	Num. days sampled	Entrainment estimate[a]	Lower 95% CI[a]	Upper 95% CI[a]
1979	3412.0	2733.3	4002.0	60	1674.7	1341.6	1964.3
1980	4365.5	3554.9	5310.5	59	2031.8	1654.5	2971.6
1981	2958.3	2452.8	3941.0	58	1610.5	1335.3	2145.5
1982	3447.3	2778.0	4760.2	55	2103.0	1693.5	2903.9
1983	5934.3	4041.2	7076.8	54	2589.3	1763.3	3087.9
1984	3482.4	2892.0	4799.6	65	2153.6	1563.9	2595.4
1985	3636.3	2524.0	5893.4	64	1897.2	1316.9	3074.8
Total					13479.8		

[a] $x10^6$ of fish larvae or eggs.

To investigate possible trends in icthyoplankton abundance during the period of two-unit operation, annual mean densities (no./500 m^3) were calculated for over 50 taxa of fish larvae at EN and NB (Appendices VI and VII) and 29 taxa of fish eggs at EN (Appendix VIII). For the selected taxa these densities are also presented in Tables 10-12. Because the standard errors of the mean densities were 20 to 200 times larger than the corresponding means, these data could not be used to describe trends. The variablity associated with these taxa as a group, appeared related to the very high variability in the abundance of anchovies (Fig. 6). Anchovy, winter flounder, sand lance, and grubby larvae and cunner, tautog, and anchovy eggs comprised over 80% of the overall mean density of larvae and eggs at EN for 1976-1985 (Table 8; Appendix V). Based on mean densities, anchovies were the most abundant larval taxon (mean density = 244/500 m^3, 61% of total density at EN and 330/500 m^3, 62% at NB). Cunner eggs were dominant (mean density = 16,281/500 m^3, 56% of total density) and little variation was seen in annual mean densities of eggs.

Table 10. Annual mean fish larval density (no./500m^3) of selected taxa at EN during 1976-1985.

Taxa	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Mean
Anchoa spp.	185.2	178.2	72.0	313.3	346.7	550.5	182.3	301.7	66.3	245.2	244.1
Pseudopleuronectes americanus	42.0	27.6	31.0	26.8	59.5	31.4	59.2	70.6	45.9	36.2	43.0
Ammodytes americanus	10.8	55.2	110.8	50.6	65.7	52.8	12.4	13.4	8.3	4.7	38.5
Myoxocephalus aenaeus	4.7	11.7	7.1	10.7	11.2	25.5	20.8	20.5	16.0	19.9	14.8
Tautogolabrus adspersus	5.6	12.1	0.4	3.2	12.4	13.8	8.6	12.5	1.0	2.8	7.2
Tautoga onitis	7.0	8.5	0.4	2.5	9.3	16.7	11.1	9.5	0.8	4.5	7.0
Menidia spp.	0.5	0.1	0.5	0.6	0.5	0.4	0.2	2.2	0.1	0.3	0.5
Gasterosteus aculeatus	0.1	0.1	0.2	0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.0	0.0	0.0
Other taxa	32.5	42.4	24.1	32.2	39.3	61.1	42.2	59.6	25.4	73.6	43.4
Total	288.4	335.9	246.5	440.0	544.6	752.2	336.8	490.0	163.8	387.2	398.5

Table 11. Annual mean fish larval density (no./500^3) of selected taxa at NB during 1979-1985.

Taxa	1979	1980	1981	1982	1983	1984	1985	Mean
Anchoa spp.	320.7	418.2	494.1	292.5	485.3	62.9	239.3	330.4
Ammodytes americanus	56.2	52.4	174.9	9.4	40.9	27.9	11.4	53.3
Pseudopleuronectes americanus	19.0	42.9	26.8	43.1	68.4	41.8	44.2	40.9
Tautogolabrus adspersus	19.6	24.9	17.6	34.4	42.4	7.9	8.0	22.1
Tautoga onitis	10.6	17.3	18.4	27.8	28.3	6.7	11.3	17.2
Myoxocephalus aenaeus	6.1	9.1	13.0	11.2	11.8	9.6	11.9	10.4
Menidia spp.	0.4	0.2	0.6	0.0	0.6	< 0.1	< 0.1	0.1
Microgadus tomcod	< 0.1	0.1	0.2	0.1	0.1	< 0.1	0.1	< 0.1
Gasterosteus aculeatus	< 0.1	0.0	< 0.1	0.0	0.0	0.0	0.0	< 0.1
Other taxa	36.5	36.0	58.4	94.6	71.6	45.1	49.0	56.0
Total	469.1	601.1	804.0	513.1	749.4	201.9	375.2	530.5

Table 12. Annual mean fish egg density (no./500m^3) of selected taxa at EN during April through September of 1979-1985.

Taxa	1979	1980	1981	1982	1983	1984	1985	Mean
Tautogolabrus adspersus	2454.5	2898.5	1908.9	1761.6	2001.2	2240.7	3016.3	2326.0
Tautoga onitis	714.6	1311.4	1273.9	1033.5	999.2	1122.0	1507.5	1137.4
Anchoa spp.	413.0	306.1	339.4	218.4	636.7	1196.9	53.2	452.0
Menidia spp.	0.0	3.4	1.5	0.3	0.7	19.8	< 0.1	3.7
Pseudopleuronectes americanus	0.8	0.9	4.7	1.1	0.3	0.0	0.0	1.1
Myoxocephalus aenaeus	3.2	< 0.1	< 0.1	0.9	< 0.1	0.0	0.5	0.6
Ammodytes americanus	0.3	0.0	0.0	0.0	0.0	0.0	0.0	< 0.1
Other taxa	377.2	138.8	277.5	253.3	220.6	235.5	374.7	268.0
Total	3963.6	4659.1	3803.9	3269.1	3858.7	4814.9	4952.2	4188.8

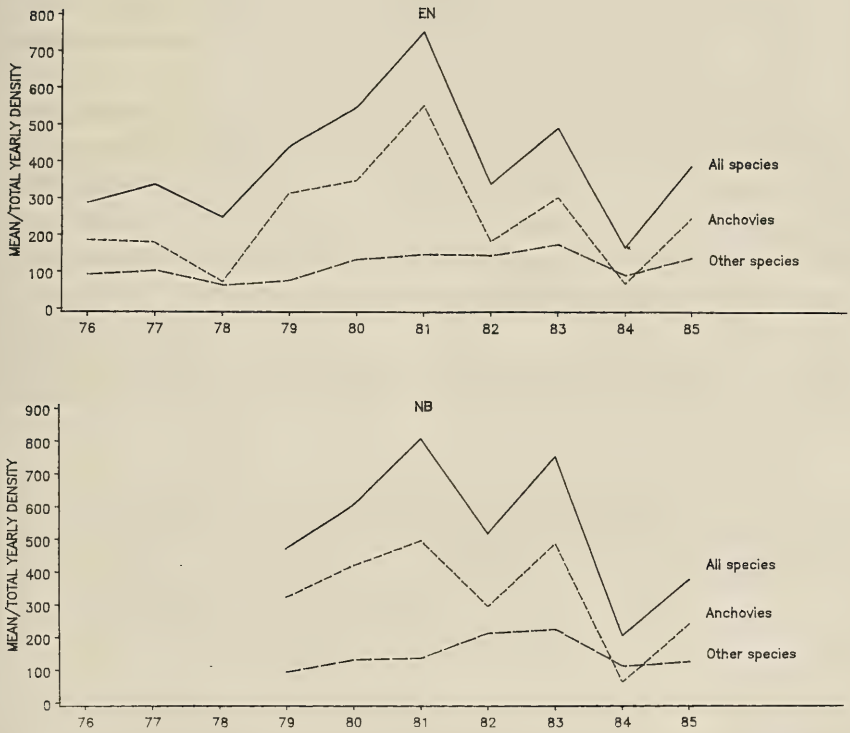


Figure 6. Annual mean density (no./500 m^3) for all larval species combined, anchovies and all other species at EN and NB.

Impingement

Impingement estimates were calculated from January 1975 to December 1985 using daily cooling water volumes (for 24 h starting at 0800 h). Over 100 taxa of fish and invertebrates have been impinged over the past 10 years (Appendix IX). Impingement estimates for the selected taxa are presented in Table 13. Sand lance accounted for almost 50% of the total number impinged. Prior to 1984, sand lance accounted for less than 1% of the total impingement (NUSCo 1986a). However, in 1984, an estimated 390,000 sand lance were impinged at Unit 2 during the week of July 18th. This estimate was based on a single 24-h sample, and qualitative observations made during the remainder of the week indicated that the numbers of impinged sand lance decreased rapidly thereafter. The sand lance is a schooling species (Leim and Scott 1966) and a large school probably encountered the intake structures. A similar short-term large impingement of a schooling species, Atlantic menhaden (*Brevoortia tyrannus*) occurred in 1971 (NUSCo 1982b). At that time, approximately one million juvenile menhaden were impinged in August at Unit 1 (the only unit operating then). Excluding sand lance, seven fish taxa dominated the impingement collections and accounted for over 90% of the fish impinged. These included winter flounder, silversides, grubby, anchovies, Atlantic tomcod, cunner, and sticklebacks. Except for the cunner and anchovies, these species were impinged in the winter months (December through March). The winter flounder was the most abundant and accounted for approximately 20% of the annual impingement catch.

There was no apparent trend for the total number of fish impinged annually during the two-unit operational period (Table 13). Although impingement samples were collected only at Unit 1 after 1983,

Table 13. Annual impingement estimates for selected taxa impinged, calculated using flows from 0800-0745 (Units 1 & 2 combined by year except during 1984-85 when Unit 2 was estimated alone).

Taxa	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
Ammodytes americanus	65	69	277	98	192	269	136	449	485411	73	487039
Pseudopleuronectes americanus	5654	7622	7676	23544	7207	7640	8875	13467	2542	2765	86992
Anchoa spp.	5606	804	869	3340	4426	4755	5895	52280	4200	342	82517
Myoxocephalus aenaeus	2108	2357	7528	3699	10736	5450	6486	14634	2359	4553	59866
Menidia spp.	1585	1328	12155	12187	10199	3733	3872	8136	1042	1480	55717
Microgadus tomcod	91	339	2398	1455	1314	8121	11868	2860	4938	1129	34513
Gasterosteus spp.	2411	5375	5511	9918	7441	.	.	.	.	.	30656
Gasterosteus aculeatus	.	.	.	.	.	6817	2951	9472	1055	852	21147
Tautogolabrus adspersus	903	1429	1862	3110	1157	2566	3851	2900	1188	466	19432
Gasterosteus wheatlandi	.	.	.	.	.	601	1393	14381	702	21	17098
Tautoga onitis	883	809	1074	866	338	814	1579	1512	664	122	8661
Other taxa	6222	7777	7187	9318	9502	11207	14530	38377	7286	4463	115893
Total	25528	27909	46517	67535	52512	51973	61436	158468	511387	16266	1019531

the highest estimated impingement occurred in 1984 due to the large number of sand lance impinged that year. Without this large impingement, the estimate (25,976) for that year would have been among the lowest since 1976. The smallest estimated impingement occurred in 1985 after sampling at Unit 1 was eliminated.

Trawls

Over 90 taxa of fish were taken by trawl at six stations in the vicinity of MNPS during the 10 yr of two-unit operation (Appendices X and XI); Oviat and Nixon (1973) and Jefferies and Johnson (1974) found that demersal fish communities in Narragansett Bay were composed of many ot the same taxa. Six fishes comprised over 80% of the trawl catch. The winter flounder was the most abundant and accounted for approximately 45% of the total catch; about one-third of the winter flounder was caught at NR. The second most abundant species was scup. Most scup were juveniles and they accounted for almost 15% of the catch. Over 40% of them were caught at NB. Biologists from CT DEP have determined that the Niantic Bay region is the primary nursery area for scup in LIS (P. Howell, pers. comm.). The windowpane accounted for over 7% of the trawl catch and over half of them were caught at BR. Anchovies, skates, and silversides had similar catches and together accounted for an additional 15% of the total. Over 90% of the anchovies were taken in Niantic Bay at IN and NB. Skates (66%) were mostly caught at the two deep-water stations (TT and BR), while silversides were often found at the nearshore stations (IN, 37%; NR, 27%; and JC, 25%).

The total annual catches of the selected taxa are presented in Tables 14 and 15. Their catch remained relatively stable, except for anchovies and cunner. The former exhibited large year to year fluctuations in catch, probably because of their highly variable distribution, and the latter showed a decline since 1976. This apparent decline of cunner is discussed later.

Table 14. Trawl catch of selected taxa by year (1976-1985).

	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
Number of samples	908	936	936	936	972	935	936	936	936	936	9367
Taxa											
Pseudopleuronectes americanus	7875	5752	6055	10694	12378	13124	13517	16799	14027	8869	109090
Anchoa spp.	980	580	2223	15	113	577	39	88	178	9997	14790
Menidia spp.	2151	1224	1060	2059	1003	356	427	635	348	465	9728
Tautogolabrus adspersus	1009	1032	359	1381	981	825	561	412	246	143	6949
Myoxocephalus aenaeus	191	276	591	316	458	866	788	904	595	498	5483
Tautoga onitis	251	292	246	283	138	235	228	159	110	136	2078
Microgadus tomcod	19	25	40	49	125	279	1147	132	90	85	1991
Gasterosteus aculeatus	19	22	13	103	38	192	116	256	940	199	1898
Ammodytes americanus	1	4	60	127	37	117	14	19	10	19	408
Other taxa	6678	7560	5855	7760	9398	15350	15310	15013	13271	10733	106928
Total	19174	16767	16502	22787	24669	31921	32147	34417	29815	31144	259343

Table 15. Trawl catch of selected taxa by station (1976-1985).

Station	JC	NR	NB	TT	BR	IN	Total
Number of samples	1563	1559	1563	1563	1563	1556	9367
Taxa							
Pseudopleuronectes americanus	10786	33899	15760	18697	12923	17025	109090
Anchoa spp.	718	240	10263	293	15	3261	14790
Menidia spp.	2186	2380	1312	493	152	3205	9728
Tautogolabrus adspersus	1412	297	495	260	377	4108	6949
Myoxocephalus aenaeus	846	2075	329	426	647	1160	5483
Tautoga onitis	442	427	252	177	222	558	2078
Microgadus tomcod	638	466	484	71	27	305	1991
Gasterosteus aculeatus	1254	612	8	10	6	8	1898
Ammodytes americanus	19	94	4	28	257	6	408
Other taxa	10943	6464	26911	17632	25901	19116	106928
Total	29205	46954	55818	38087	40527	48752	259343

Seines

Approximately 30 different taxa have been caught by seine during the 10 yr of two-unit operation (Appendices XII and XIII). Hillman et al. (1977), found many of the same taxa in the MNPS area. Two taxa accounted for over 90% of the total seine catch from 1976 through 1985. Silversides dominated the shore-zone seine catches and accounted for about 75% of the total; *Fundulus* spp. (mummichogs and striped killifish) accounted for an additional 12%. About 80% of the total seine catch (Table 16) was

Table 16. Seine catch of selected taxa by station (1976-1985).

	JC	WP	GN	Total
Number of samples	300	314	321	935
Taxa				
Menidia spp.	72783	9879	7817	90479
Ammodytes americanus	2	197	855	1054
Gasterosteus aculeatus	227	19	23	269
Pseudopleuronectes americanus	19	7	59	85
Gasterosteus wheatlandi	12	13	14	39
Myoxocephalus aenaeus	4	7	7	18
Anchoa spp.	11	1	0	12
Tautogolabrus adspersus	5	1	0	6
Tautoga onitis	4	0	0	4
Other taxa	12953	2517	1401	16871
Total	86020	12641	10176	108837

Table 17. Seine catch of selected taxa by year (1976-1985).

	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
Number of samples	66	72	72	72	72	72	80	99	174	156	935
Taxa											
Menidia spp.	40620	18179	1178	1233	7764	3418	5408	9007	2330	1342	90479
Ammodytes americanus	6	520	16	51	10	318	82	30	21	0	1054
Gasterosteus aculeatus	8	151	8	30	2	3	5	49	9	4	269
Pseudopleuronectes americanus	4	6	4	1	2	9	2	1	18	38	85
Gasterosteus wheatlandi	.	.	.	.	.	8	3	5	5	18	39
Myoxocephalus aenaeus	3	2	1	2	0	0	3	1	3	3	18
Anchoa spp.	0	0	0	0	2	0	7	2	1	0	12
Tautogolabrus adspersus	0	0	2	0	0	0	3	0	1	0	6
Tautoga onitis	0	0	0	0	0	0	4	0	0	0	4
Other taxa	2302	2466	1200	1116	1051	1114	1254	3112	2117	1139	16871
Total	42943	21324	2409	2433	8831	4870	6771	12207	4505	2544	108837

taken at JC. Hundreds of juvenile silversides were routinely caught at this site during the summer months because this station is in a nursery area for shore-zone fish.

Total annual seine catches of all taxa combined (Table 17) showed no apparent trend during the period, although totals were highest in 1976 and 1977. Because silversides dominated all the annual catches, total catches were largely a function of silversides catches. Over half of all the silversides were caught in 1976 and 1977, thus catches for all taxa combined were also highest in 1976 and 1977. Except for silversides, there has been no apparent trend in the total annual seine catches of the selected taxa since 1976. Results of the silversides data analysis are discussed later.

Selected species

Ammodytes americanus, American sand lance

The American sand lance (*Ammodytes americanus*) is found from the Arctic to Cape Hatteras (Bigelow and Schroder 1953). They are primarily pelagic plankton feeders (Richards 1982). Individuals form large schools and are found over sandy bottoms from near shore to the edge of the continental shelf (Richards 1963; Leim and Scott 1966). Sand lance mature in one to two years and spawn between December and March (Westin et al. 1979). Covill (1959) and Meyer et al. (1979) reported that large annual fluctuations of sand lance abundance occur along the Atlantic coast.

Sand lance have been collected in all fish programs. Except for 1984, when a large school of sand lance was impinged, these fish have generally contributed less than 1% to annual total impingements (NUSCo 1984a). The sand lance is a winter spawner and its larvae were collected from January to May (Fig. 7); its abundance ranked third at EN and second at NB (Tables 10 and 12). Annual entrainment estimates during the two-unit operation period (1976-1985), were based on median densities and ranged from 5.8×10^6 (1985) to 66.7×10^6 (1977) (Table 9). Because sand lance eggs are demersal and adhesive (Frizsche 1978), they were rarely collected. Sand lance were collected infrequently in the trawl and seine samples, probably because juvenile and adult sand lance burrow into the sand (Leim and Scott 1966), thereby avoiding these gear.

Annual impingement estimates for sand lance never exceeded 450 except in 1984 when 390,000 sand lance were impinged during the week of July 18 (NUSCo 1985). This mass impingement of sand lance did not recur and the estimated number of sand lance impinged in 1985 was comparable to historic levels (Table 13). Time-series analysis did not adequately describe the fluctuations of impinged sand lance because of the large numbers in 1984 and the low levels of impingement during all the other years.

Larval sand lance were abundant in plankton collections from 1977-1981, but a marked decrease in densities occurred after 1981 (Fig. 7). The temporal catch distribution of larval sand lance was as variable as that seen in impingement (Tables 10 and 12). Larval sand lance were caught in large numbers during short time periods. For example, more than 60% of the annual 1978 cumulative density was caught at EN between January 23 and February 3, and more than 35% of the 1980 cumulative density on May 19.

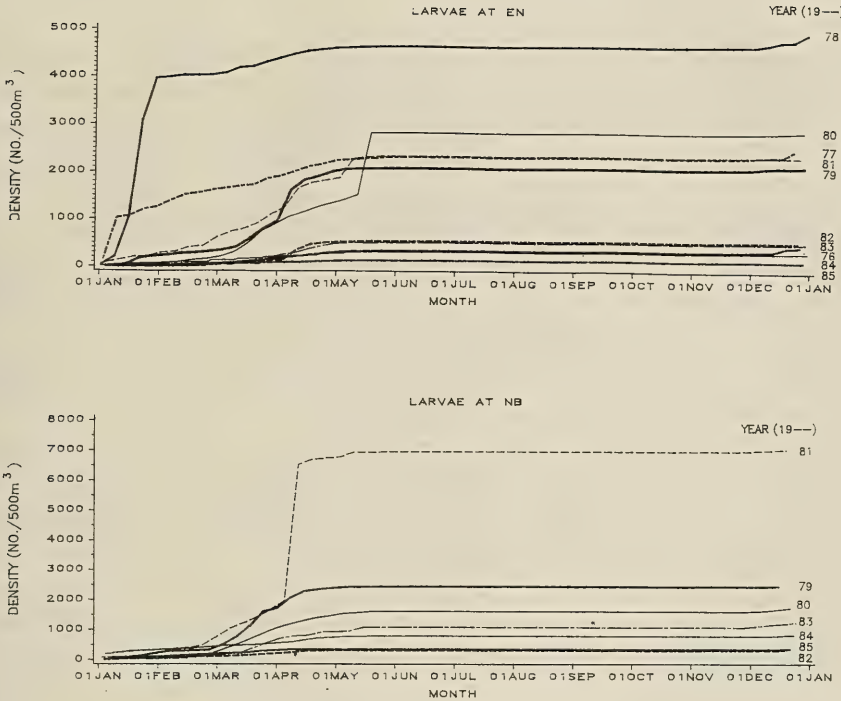


Figure 7. Annual cumulative density (no./500 m^3) of sand lance larvae at EN and NB.

At NB, more than 70% of the 1981 cumulative density was collected on April 15. These patterns could be explained by the finding that larvae that hatch together tend to remain together (Norcross et al. 1961).

Time-series models of larval density at EN and NB fitted observed data well and had R^2 values of 0.74 and 0.92, respectively (Fig. 8; Appendix XIV). These models included periodic components of 4 and 6 mo, suggesting that the abundance of sand lance larvae is inherently seasonal within each year. The previously mentioned decline in abundance of sand lance after 1981 has become a feature of the time-series model, which provides an average characterization of the years of high (1977-1981) and low (1982-1985) abundance. Therefore, the measures of variation from the model forecasts (MSE) at EN and NB (Appendix XIV) are presented as a two-unit operational baseline for future assessments.

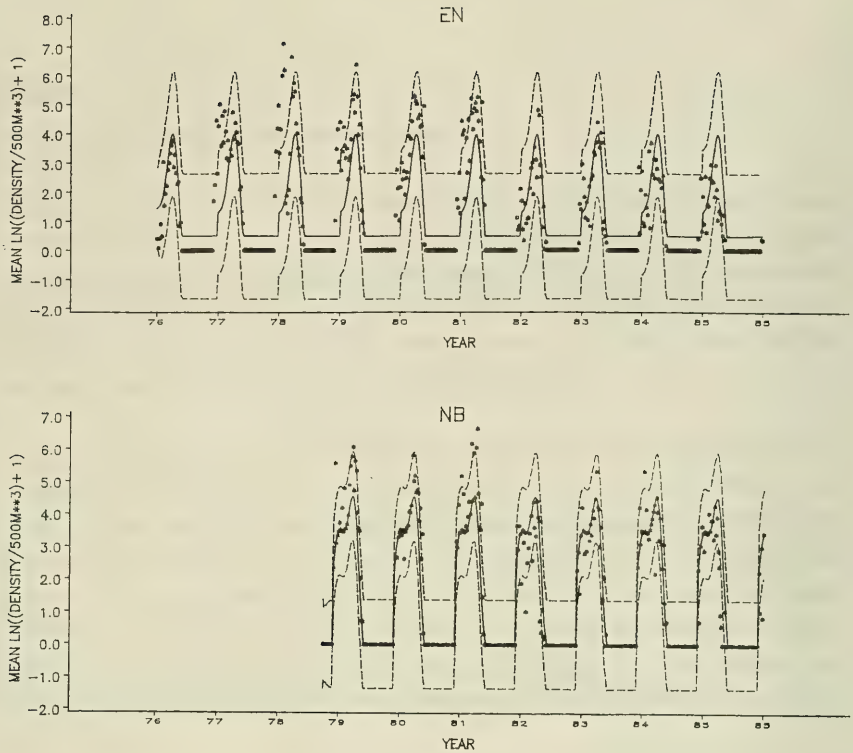


Figure 8. Time series plots of sand lance larvae at EN and NB; forecast (—), 95% confidence limits (---), and mean weekly log-transformed density (no./500 m^3) (*).

Anchoa spp., anchovies

Two anchovy species, the bay anchovy (*Anchoa mitchilli*) and the striped anchovy (*Anchoa hepsetus*) have been collected in the MNPS area. Eggs of the two species can be easily distinguished and, based on their relative proportions, the bay anchovy was by far the most common at MNPS (more than 80%).

The bay anchovy is perhaps the most abundant fish along the Atlantic Coast (McHugh 1977). They are commonly found inshore during the warmer months and move offshore in the winter. Hildebrand (1943) believed that each section of the coast had a distinctive population and all migrations were inshore

and offshore movements. In LIS, spawning takes place in depths of less than 20 m during June through September (Richards 1959). Eggs are pelagic and hatch in 24 h at approximately 27°C (Kuntz 1914). Development is rapid and individuals may mature within 2.5 mo of hatching, at a size of 34 to 40 mm; its life span is probably not more than 2 or 3 yr (Stevenson 1958).

Anchovies were among the four most abundant taxa collected in all programs except seines. Larval abundance ranked first at both EN and NB and egg abundance ranked third at EN (eggs from NB were not identified) (Appendices VI, VII and VIII). During the two-unit operational period, annual larval entrainment estimates, based on median densities ranged from $1.5x10^6$ in 1984 to $1,284.1x10^6$ in 1981. Although median larval densities at EN were generally low (1976-1978), high (1979-1981), and low again (1982-1985), the actual number of anchovies entrained during the same periods did not follow that pattern because entrainment is a function of plant operating conditions. However, the highest and the lowest estimates at EN also occurred in 1981 and 1984, respectively. In addition, both the entrainment estimate and median density of larval anchovies at EN in 1981 were significantly higher ($p < 0.05$) than in any other year (Table 9). Also the number of larval anchovies entrained in 1984 was significantly lower than in any other year. Annual egg estimates ranged from $16.0x10^6$ in 1985 to $807.7x10^6$ in 1984 (Table 9). Although annual impingement estimates varied more than two orders of magnitude (342 to 52,280), it was the third most abundant taxon impinged (Appendix IX). Among the species taken in trawls, anchovies ranked fourth (Appendices X and XI).

Near MNPS, anchovies migrated inshore in May and June and were available to the various NUEL sampling gear through October (Fig. 9). Adults (median length of 77 mm) were impinged primarily from May through June, which corresponded to the time of their spawning. Eggs were abundant from June through July and larvae July through August at EN and NB. Juvenile anchovies (median length of 27 mm) were caught in trawls during August through October, primarily at NB and IN.

Because anchovies mature in one year, changes in larval density in any given year should result in corresponding changes in adult and egg abundances the next year. However, this pattern was clearly not observed in the annual catches of anchovies during the last three years (1983-1985) of the two-unit operational period. In 1983 and 1984, the densities of eggs at EN and the number of adults impinged were equal to or higher than in previous years, but the 1984 larval densities at both EN and NB were lower than in any previous year (Tables 10, 11, and 13; Fig. 10). This minimum was probably related to

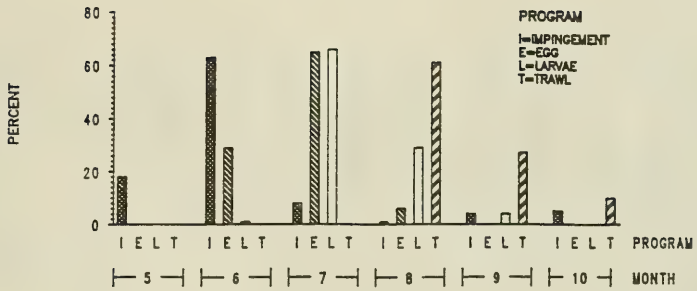
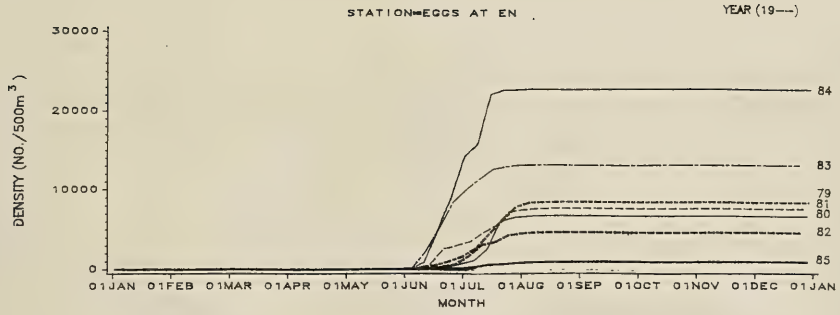
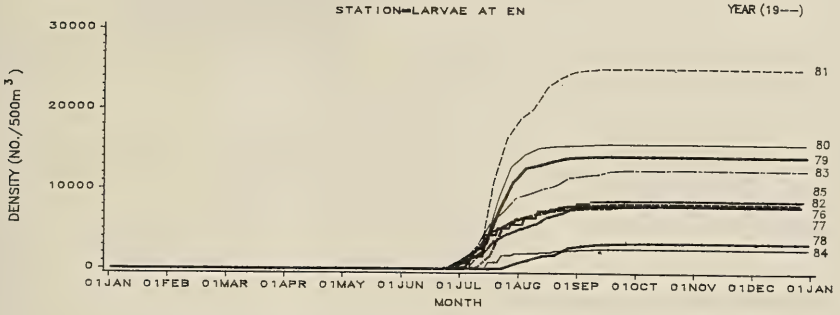


Figure 9. The monthly percent distribution of anchovies collected in impingement plankton and trawl samples.

the high abundance of a planktivorous ctenophore that occurred during the summer of 1984 (see tautog subsection for a further discussion of this phenomenon). As expected, the catch of adults in impingement samples and the density of eggs decreased the following year. However, later in 1985, a large increase occurred in larval density (compared to 1984), which was followed by the largest annual catch of young-of-the-year in trawls (Fig. 11). Because the 9.1-m trawl does not sample anchovies well, due to their small size and patchy distribution, year to year comparisons based on these data are generally not useful. Nevertheless, the 1985 increase was probably real because the mean annual trawl catch for that year was nearly an order of magnitude larger than in any other year, and was significantly larger than the mean in any of the previous three years. These fluctuations did not appear to be related to MNPS operations nor to the abundance of adults and eggs, but rather caused by density-dependent mechanisms as other researchers have reported (Lasker 1974).

The seasonality of anchovy catches in plankton and impingement programs was described by the periodic components in the time-series models (Appendices XV, XVI, and XVII). The models of egg and larval abundance explained the data better (R^2 values larger than 0.84) than the impingement model did ($R^2 = 0.69$). The 1984 decline in larval abundance is now a feature of the two-unit operational series described by the model (Appendix XVII). The lower R^2 value of the impingement model was probably the result of the unusually high 1983 catches (NUSCo 1984a). Summaries of these baseline models are presented in Appendices XV, XVI, and XVII.





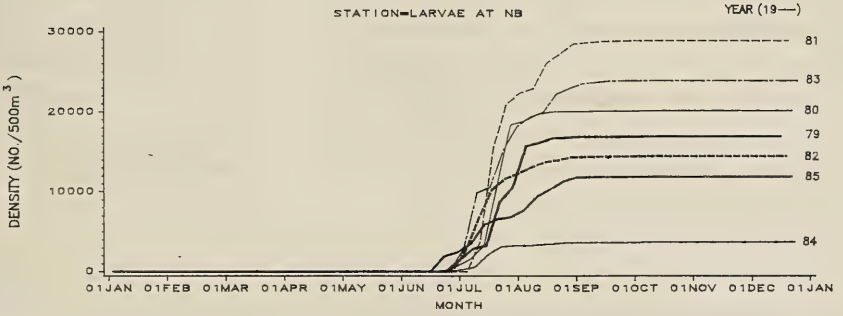


Figure 10. Annual cumulative density (no./500 m^3) of anchovy eggs at EN and larvae at EN and NB.

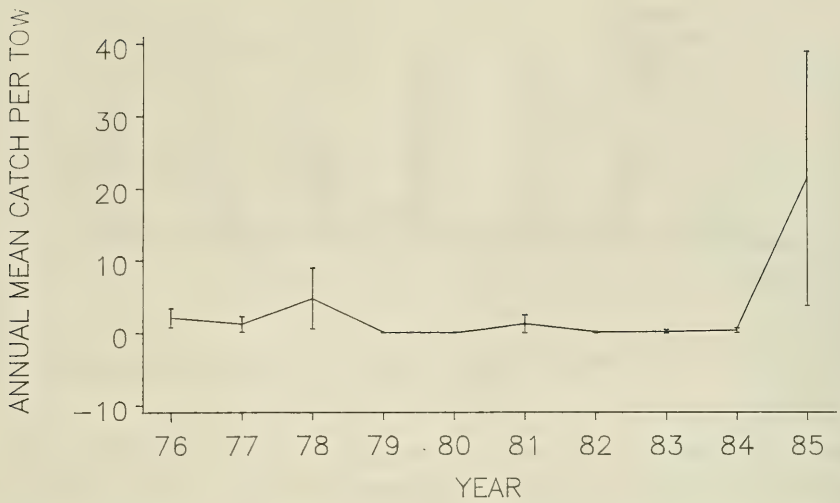


Figure 11. Annual mean catch of anchovies taken by trawl; the vertical bars are approximated 95% confidence intervals for each year.

Gasterosteus spp., sticklebacks

The threespine stickleback (*Gasterosteus aculeatus*) and the blackspotted stickleback (*Gasterosteus wheatlandi*) are small, nearshore fishes. The threespine stickleback is distributed throughout the north polar regions and as far south as Chesapeake Bay in the Western North Atlantic. The blackspotted stickleback is found only in the Western North Atlantic from Newfoundland to LIS (Perlmutter 1963). During the spring, both species move into salt marshes and tidal rivers to spawn (Worgan and Fitzgerald 1981). However the two species are found in different salinity regimes during their reproductive season, and thus do not compete for resources during spawning (Audet et al. 1985). The threespine stickleback is the larger fish of the two species.

Threespine and blackspotted sticklebacks are very similar in appearance and are not easily distinguished (Bigelow and Schroeder 1953). Because of this similarity, the blackspotted stickleback was not identified in MNPS collections until October 1981 (NUSCo 1982a). Although Fitzgerald and Whoriskey (1985) found no size overlap between these two species, the length frequency distributions of individuals collected

at MNPS overlapped at 30 to 35 mm (Fig. 12). Thus, length frequencies could not be used to separate the species in earlier data. Data for the two species were combined.

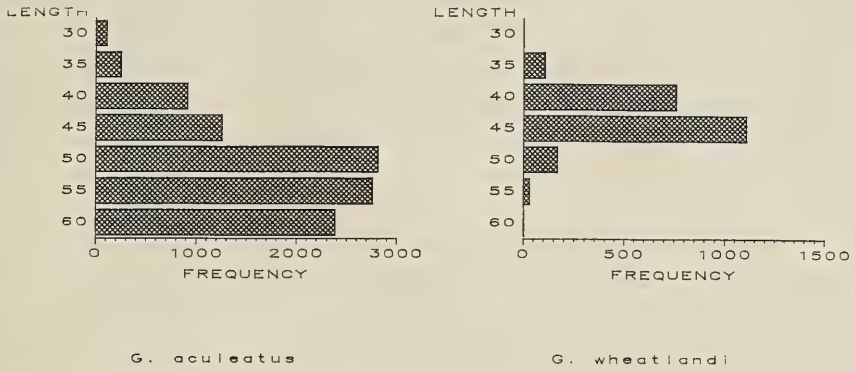


Figure 12. Length frequency distribution of sticklebacks.

Blackspotted and threespine sticklebacks were collected in all programs, but were only abundant in impingement samples from fall through spring. Adults and young-of-the-year remain in the spawning areas until late summer (Fitzgerald 1983), which may account for the low abundance of these fish during the summer. Approximately 35% of all sticklebacks impinged at MNPS were taken in 1983 (Table 13). The estimated impingment of sticklebacks in 1984 and 1985 was much lower, because sampling had been eliminated at Unit 1. The estimated number of sticklebacks impinged ranged from 2,411 to 9,918 in the period of two-unit operation prior to 1984, but no trend was apparent and, presumably, no impact had resulted from MNPS operations.

Sticklebacks were sufficiently abundant for time-series modeling in only impingement samples. The R^2 for the impingement time-series model was 0.82, demonstrating a good fit to the data. Cooling-water flow, an annual periodic component, and a short term 4-mo cycle described impinged stickleback abundance. A summary of this baseline model is presented in Appendix XVIII.

Menidia spp., silversides

Two species of silversides dominate the shore zone along the Connecticut coast, the Atlantic silverside (*Menidia menidia*) and the inland silverside (*Menidia beryllina*). Both species are sympatric along the Atlantic coast, with the Atlantic silverside ranging from the Gulf of St. Lawrence to the Chesapeake Bay and the inland silverside ranging from Cape Cod to South Carolina (Johnson 1975). Both species spawn as yearlings and have a life cycle that ranges from one to two years. Both are omnivorous, feeding on copepods, mysid shrimp, fish eggs and young squid. They are important as forage food for larger fish species (Bigelow and Schroeder 1953).

Silversides collected in MNPS programs were identified to genus during some portion of the two-unit operational period. When identified to species, *Menidia menidia* were the most abundant (over 90%). However, to investigate long-term trends the two species were always analyzed together as a single taxon. Silversides, the most abundant taxon among those collected in the seine program, ranked third and fifth among taxa sampled by the trawl and impingement programs, respectively. Silversides were not abundant in plankton samples because their eggs are adhesive (Bigelow and Schroeder 1953) and larvae and juveniles stay close to shore (Bayliff 1950).

Pronounced seasonal patterns of abundance were evident in seine, trawl and impingement collections (Fig. 13). The seine catches in the summer and early fall were composed primarily of juveniles (20-50 mm), while catches in trawl and impingement collections were composed of larger fish (60-120 mm) collected from deeper waters in the winter. This pattern of catches seems related to the offshore migrations of silversides in the late fall and winter reported in other studies (Bayliff 1950; Bigelow and Schroeder 1953; Conover 1979). The average monthly catches of silversides taken by trawls in the MNPS area during the last 10 yr also showed a pattern pointing out to regular winter offshore migrations (Fig. 14). Silversides began to move away from the shore-zone and were found at the nearshore station, JC, in September. They were collected from NR, NB and IN in November and were found at the two offshore stations, BR and TT, December through March.

Although the total annual seine and trawl catches of silversides (Tables 17 and 14, respectively) suggest a common and decreasing trend, a similar pattern was not evident among annual impingement estimates (Table 13). Further, the variances associated with those data were high as demonstrated by the

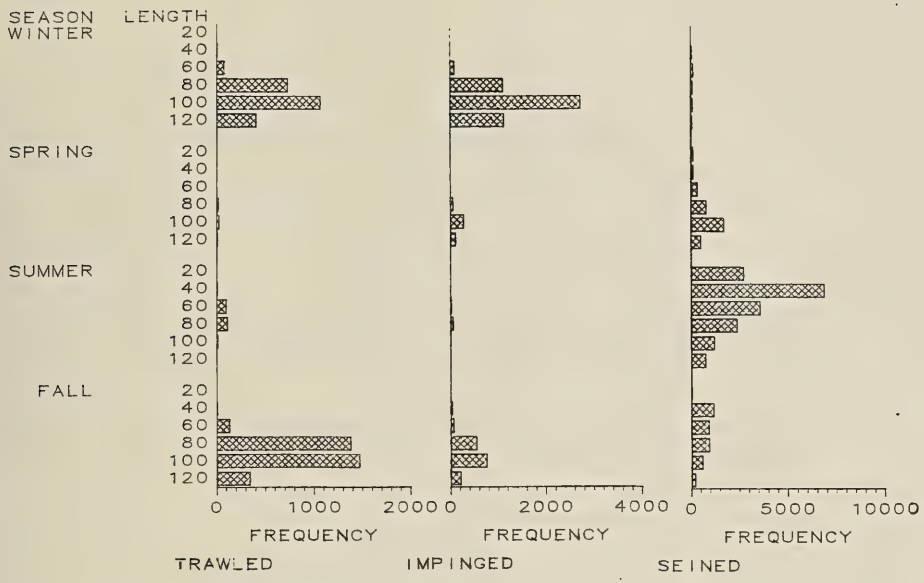


Figure 13. Length frequency distribution of silversides by season.

wide (and overlapping) 95% confidence intervals in Figure 15. Because NUSCo (1984b) found that catches from GN, a control station located away from the influence of MNPS, were correlated with catches from WP, a potentially impacted station, the apparent declines in both seine and trawl catches were attributed to a regional decline unrelated to the construction and operation of MNPS.

Silversides abundance was well described by the time-series models for seine data ($R^2 > 0.80$; Appendix XIX) and impingement data ($R^2 = 0.75$; Appendix XX). All models had an annual periodic component, except the seine model for JC station which had only a 6-mo period. Summaries of these baseline models are presented in Appendices XIX and XX.

Microgadus tomcod, Atlantic tomcod

The Atlantic tomcod (*Microgadus tomcod*) is the most abundant member of the cod family collected in the monitoring programs at MNPS. It ranges along the Atlantic coast of North America from

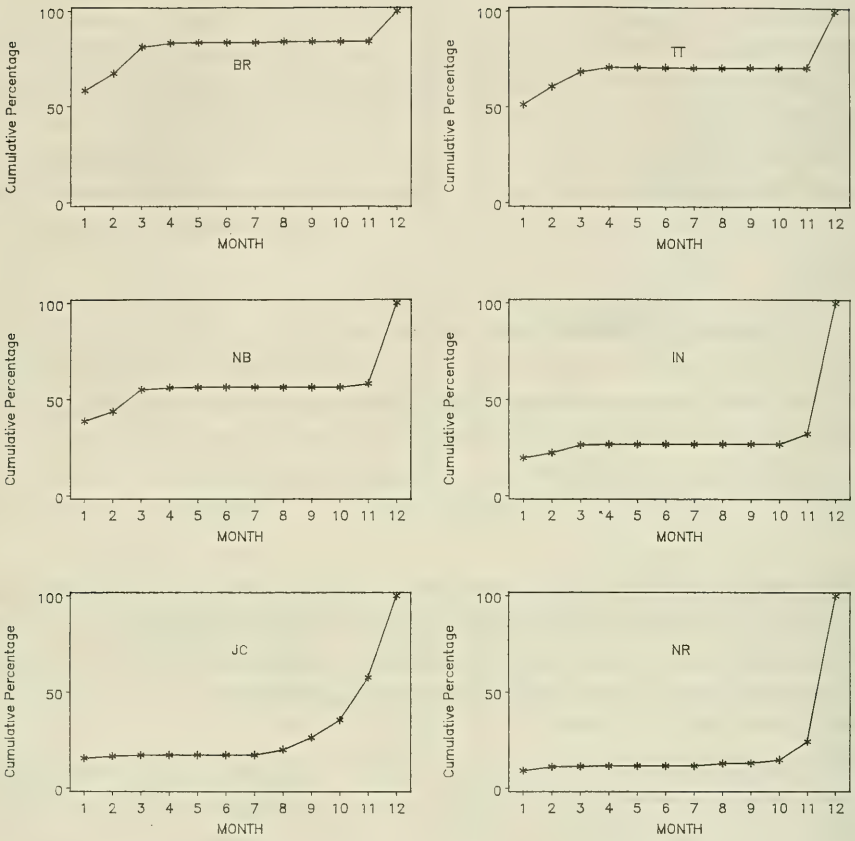


Figure 14. Silversides 10-yr trawl-catch monthly averages (no./0.69 km) expressed as a cumulative percentage of total annual catches.

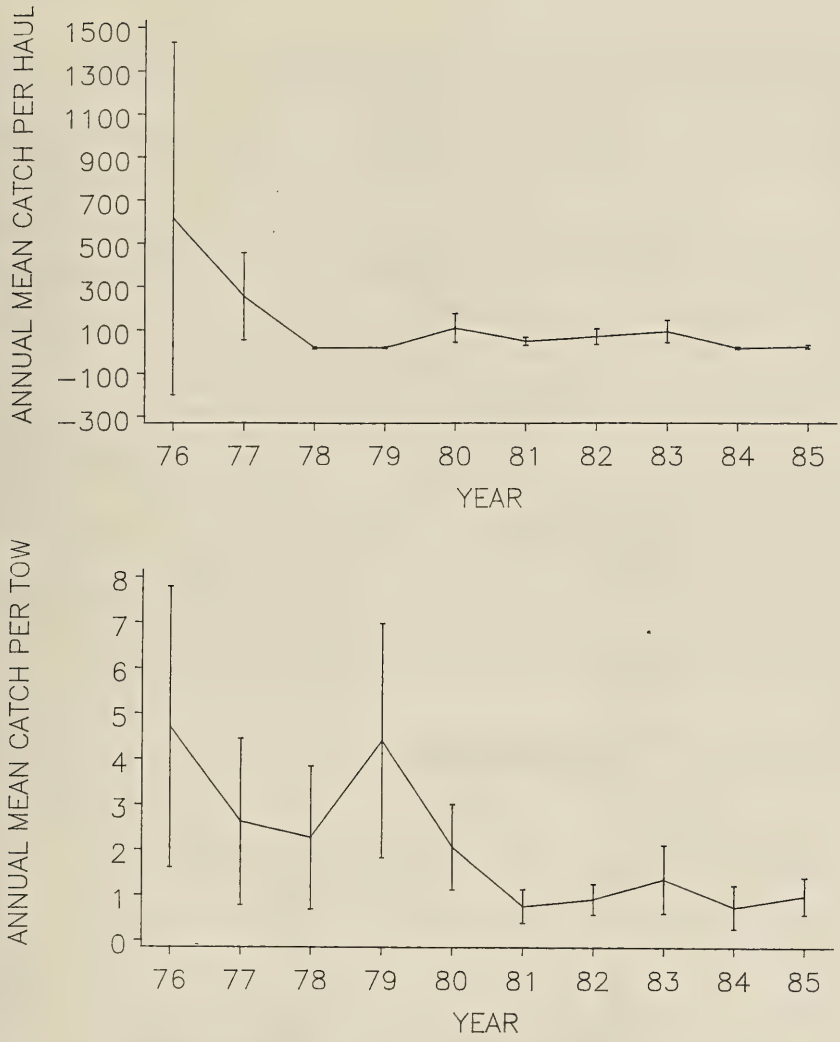


Figure 15. Silversides annual mean catches taken by seine (a) and trawl (b); the vertical bars are the approximated 95% confidence intervals for each year.

Newfoundland to Virginia (Bigelow and Schroeder 1953). Howe (1971) reported that tomcod reach sexual maturity at about 130 mm; they migrate up rivers to spawn in fresh or brackish water from November through February. Eggs are adhesive and are found attached to the substrate. After spawning, adult and larval tomcod remain in or near the estuary. They move to cooler waters during the summer months.

Tomcod were caught in all fish programs, but were more abundant in the impingement and trawl samples than in plankton collections. Eggs are adhesive and larvae tend to remain in or near spawning areas, which are habitats that were not sampled by NUSCo programs. Over 98% of impinged tomcod were adults (fish larger than 130 mm) and about 90% of these were taken during fall and winter (Fig. 16). About 58% of the total tomcod impingement estimate during the two-unit operational period, were impinged during 1981 and 1982; estimates for other years ranged from 91 to 4,938 fish (Table 13). In trawls, more individuals were caught nearshore (stations NR, IN, NB and JC) than offshore (BR, TT) (Table 15). Young-of-the-year dominated the catches of tomcod taken by trawl in the spring and summer; adults were caught mostly in the fall and winter (Fig. 16). Trawl catches were seasonal and peaked from April through June except at NR, where most fish were caught during their spawning season (Fig. 17). Tomcod trawl catches were higher in 1981 and 1982 than in any other year (Table 14); during these two years impingement estimates were also highest. This 1981-1982 peak could not be related to MNPS operation or construction activities and, thus, it was attributed to an unusually large year-class.

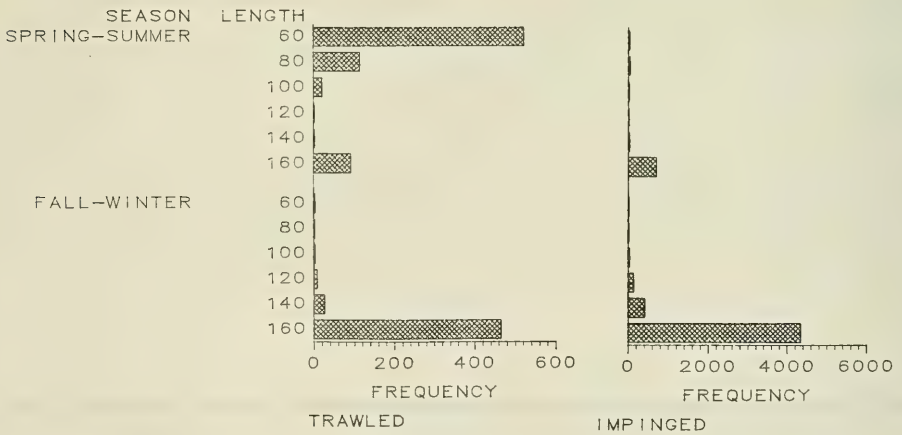


Figure 16. Length frequency distribution of tomcod.

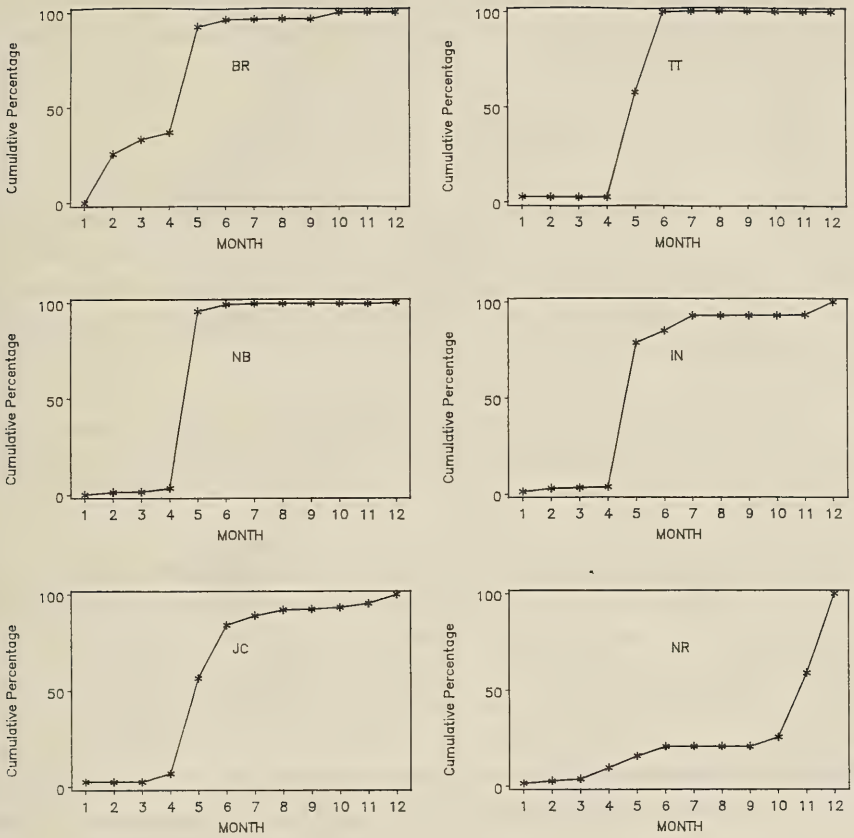


Figure 17. Atlantic tomcod 10-yr trawl-catch monthly averages (no./0.69 km) expressed as a cumulative percentage of total annual catches.

The tomcod impingement data were well described ($R^2 = 0.69$; Appendix XXI) by a time-series model. This model had annual and six-month periodic components. A summary of this baseline model is presented in Appendix XXI.

Myoxocephalus aenaeus, grubby

The grubby (*Myoxocephalus aenaeus*) is found in coastal waters, commonly in eelgrass habitats, along the Atlantic coast of North America from the Gulf of St. Lawrence to New Jersey (Bigelow and Schroeder 1953). It spawns throughout the winter (Lund and Marcy 1975) and Richards (1959) reported finding larvae in shallower areas of LIS from February to April. The grubby tolerates a wide range of temperatures and salinities (Bigelow and Schroeder 1953).

The grubby is a resident of the waters near MNPS and both larvae and adults have been collected in the NUSCo monitoring programs. Eggs were rarely collected because they are demersal and adhesive. Adult fish were also rare in seine samples.

Overall mean larval density of grubby during the two-unit operational period ranked fourth at EN and sixth at NB (Appendices VI and VII). These larvae were collected from February through May at both stations (Fig. 18). Annual entrainment estimates ranged from 9×10^6 in 1978 to 50×10^6 in 1983 and peak larval abundance, as measured by both median (Table 9) and mean densities (Tables 10 and 12), occurred in 1981. However, the temporal pattern described by the annual means and medians did not correspond to the pattern described by the cumulative mean densities shown in Figure 18. Cumulative density at NB was highest in 1985, followed by 1981, 1983 and 1982, while at EN it was highest in 1981, followed by 1982, 1983 and 1985. Most likely, these discrepancies were not related to any real differences but were a result of of the high variablity inherent of the plankton data (variances were 20 to 200 times larger than the corresponding means).

Over 90% of the grubby in the impingement and trawl samples were adults (60 to 120 mm) (Fig. 19). Adult catches ranked fourth among impinged taxa and ninth among species taken in trawls (Appendices IX and X). Impingement estimates ranged from 2,108 in 1976 to 14,634 in 1983, and 42% of the total impingement during the two-unit operational period (1976-1985) occurred in 1980 and 1983. The

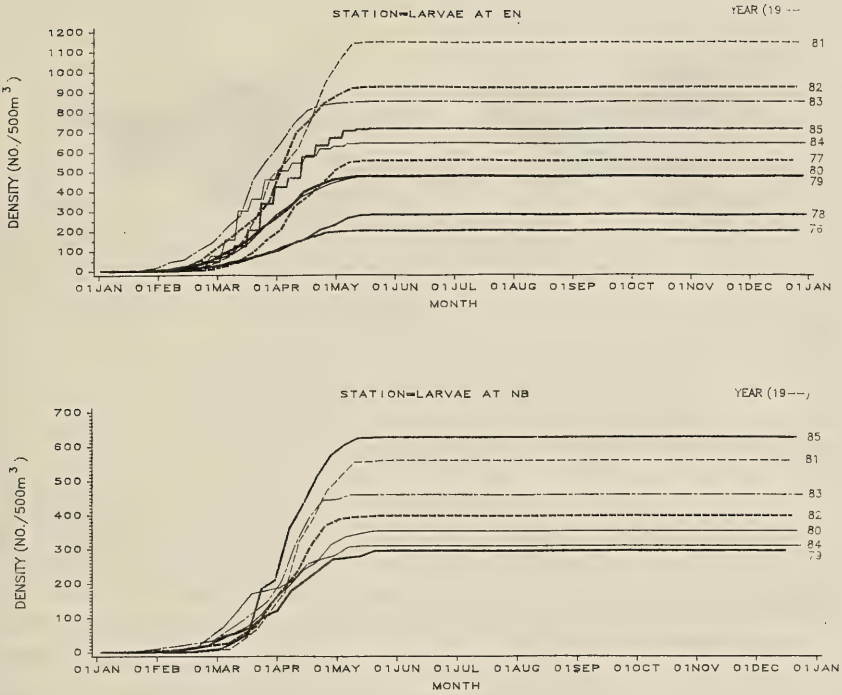


Figure 18. Annual cumulative density (no./500 m^3) of grubby larvae at EN and NB.

impingement estimates for 1984 and 1985 were relatively low and did not include Unit 1 impingement because the sluiceway was operational, but those estimated for 1976 and 1977 were smaller.

The grubby was taken in trawls primarily from November through April except at JC where it was taken throughout the year (Fig. 20). Almost 75% were caught at the nearshore stations JC, NR and IN (Table 15). Total catch of grubby taken in trawl for the two-unit operational period was 5,483 fish (Table 14) and annual catches ranged from 191 to 866. Annual mean catches did not show any trend over the 1976-1985 period (Fig. 21) and the observed fluctuations could not be related to MNPS two-unit operations.

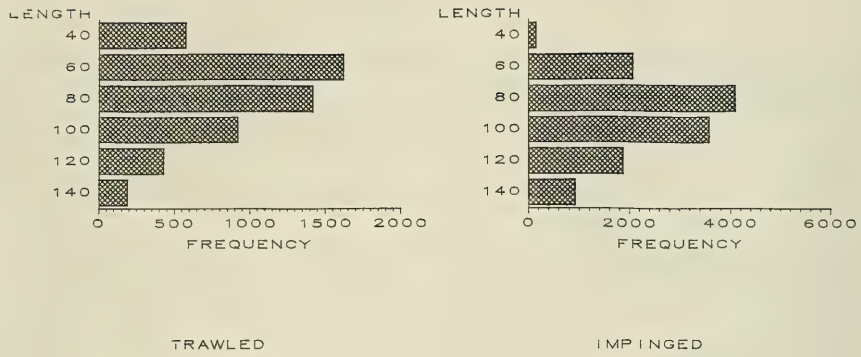


Figure 19. Length frequency distribution of grubby

Time-series models described well the temporal abundance of grubby in impingement and larval collections (Appendices XXII and XXIII). All R^2 values were over 0.90 in the larval series models. These larval models had annual and 4-mo periodic components as their main deterministic features. Cooling-water flow and an annual cycle accounted for 83% of the variability in the impingement model. All the models have been consistent for the past 3 yr (NUSCo 1984a, 1985, 1986a) suggesting that they included the most relevant variables for describing the natural fluctuations of this species. Summaries of these baseline models are presented in Appendices XXII and XXIII.

Tautoga onitis, tautog

The tautog (*Tautoga onitis*) is found from New Brunswick to South Carolina, but is most common from Cape Cod to Delaware Bay (Cooper 1965). Adult and juvenile tautog are found around rocky areas, ledges, mussel beds, breakwaters, an other similar nearshore habitats from early May until late October (Bigelow and Schroeder 1953; Cooper 1965). Juveniles are also found in eelgrass beds and among macroalgae in coves and channels (Tracy 1910; Briggs and O'Conner 1971). Both juveniles and adults have a home site where they remain inactive and under cover at night; during the day larger fish move to other locations to feed, but juveniles remain close to their home sites (Olla et al. 1974). During winter, adults move to deeper water and remain inactive while juveniles stay inshore to overwinter in a torpid state (Cooper 1965; Olla et al. 1974). Tautog males become sexually mature at age 3 and females at age 4 (Chenoweth 1963). Spawning occurs from mid-May until mid-August in LIS (Wheatland 1956;

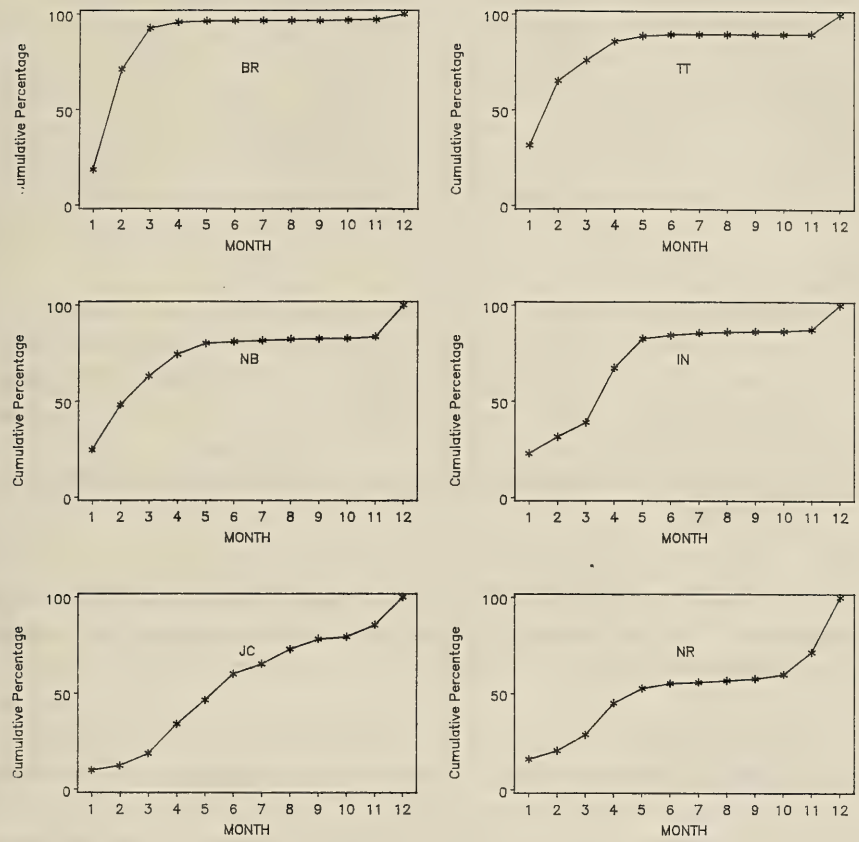


Figure 20. Grubby 10-yr trawl-catch monthly averages (no./0.69 km) expressed as a cumulative percentage of total annual catches.

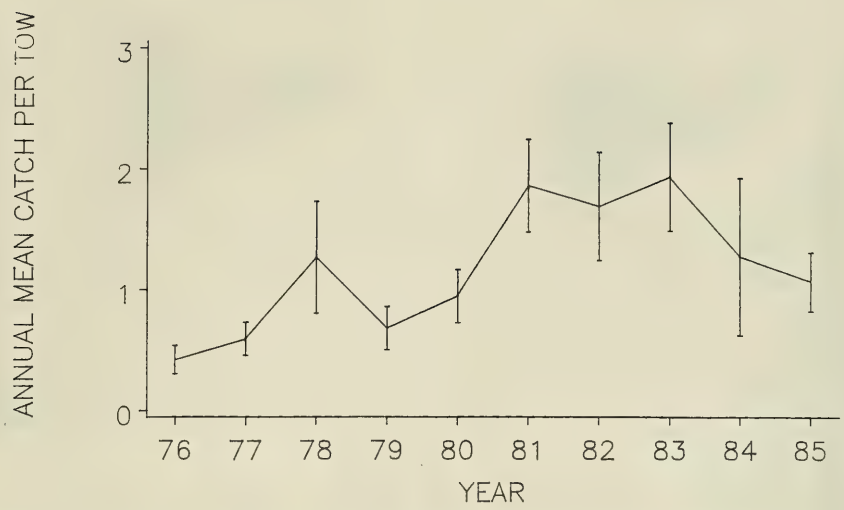


Figure 21. Annual mean trawl-catches of grubby; the vertical bars are the approximated 95% confidence intervals for each year.

Chenoweth 1963). The eggs are pelagic and are concentrated in the upper 5 m of the water column (Williams 1967). Young become benthic and move inshore after metamorphosis, which is completed by 10 mm (Fritzsche 1978).

Tautog was found in all sampling programs, but was most abundant in the June through August plankton collections (Tables 10, 12, and 14). Catches were low in impingement, trawl and seine samples.

Tautog eggs were the second most abundant egg taxon entrained (Appendix V) during the two-unit operational period. Annual egg entrainment estimates did not vary widely and ranged from 646×10^6 (1979) to about $1{,}400 \times 10^6$ (1981 and 1982). Both median and mean egg densities (Tables 9 and 11) varied over a relatively small range; medians from 920 (1979) to 1,619/500 m^3 (1981) and means from 715 (1979) to 1,506 /500 m^3 (1985). The lowest egg density as indicated by both measures occurred in 1979, but the highest egg density occurred in either 1981 or 1985, depending on the index of abundance used. Clearly, no trend can be discerned from these data.

Tautog larvae ranked fifth at NB and seventh at EN and the annual cumulative larval densities were usually greater at NB than at EN (Fig. 22). Although tautog egg abundance was relatively stable throughout the study period, larval abundance was not. Larval densities were considerably lower in 1978 and 1984 than in any other year (Tables 10 and 12). In July 1984 a large decline in the abundance of other plankton (including larval anchovies and cunner) was noted. The reasons for this 1984 decrease, which was observed in other parts of eastern LIS (Richards, per. comm.), were not known, but during July 1984, ctenophores, a plankton predator (Denson and Smayda 1982) were unusually abundant; a similar phenomenon may have occurred in 1978.

Juvenile and adult tautog were present throughout the year in both impingement and trawl collections but catches were higher May through October. Even though tautog prefer rocky shores such as those surrounding MNPS intakes, they were not impinged in large numbers and contributed less than 1% to the estimated impingement total (Table 8). Annual impingement estimates varied an order of magnitude, from 122 in 1985 to more than 1,500 in both 1982 and 1983; the low estimate in 1985 may have been related to a Unit 2 shut-down in June. More tautog were taken by trawl at nearshore stations (JC, IN and NR) than at mid-bay (NB) and offshore (TT, BR) stations. Juveniles (fish smaller than 80 mm) were taken by trawl primarily at NR (Fig. 23), which was an ideal nursery area (P. Briggs, per. comm.). Smaller adult tautog probably stayed near the rocks and shoreline and thus were more susceptible to impingement (Fig. 23). For similar reasons, these smaller fish were also abundant at JC. Because larger tautog move to open water to feed, these individuals would be more likely to be taken by trawl at BR, IN, NB and TT.

Because the numbers of tautog caught in trawl and seine samples were generally low and not representative of their temporal distribution, the indices of egg and larval density were used for impact assessment. Time-series models were fitted to the egg and larval data from the NUSCo plankton program. All models had R^2 values larger than 0.80 and included an annual cycle. The 1984 decline in larval abundance is now a feature of the two-unit operational baseline models for both EN and NB. Summaries of these models are presented in Appendices XXIV and XXV.

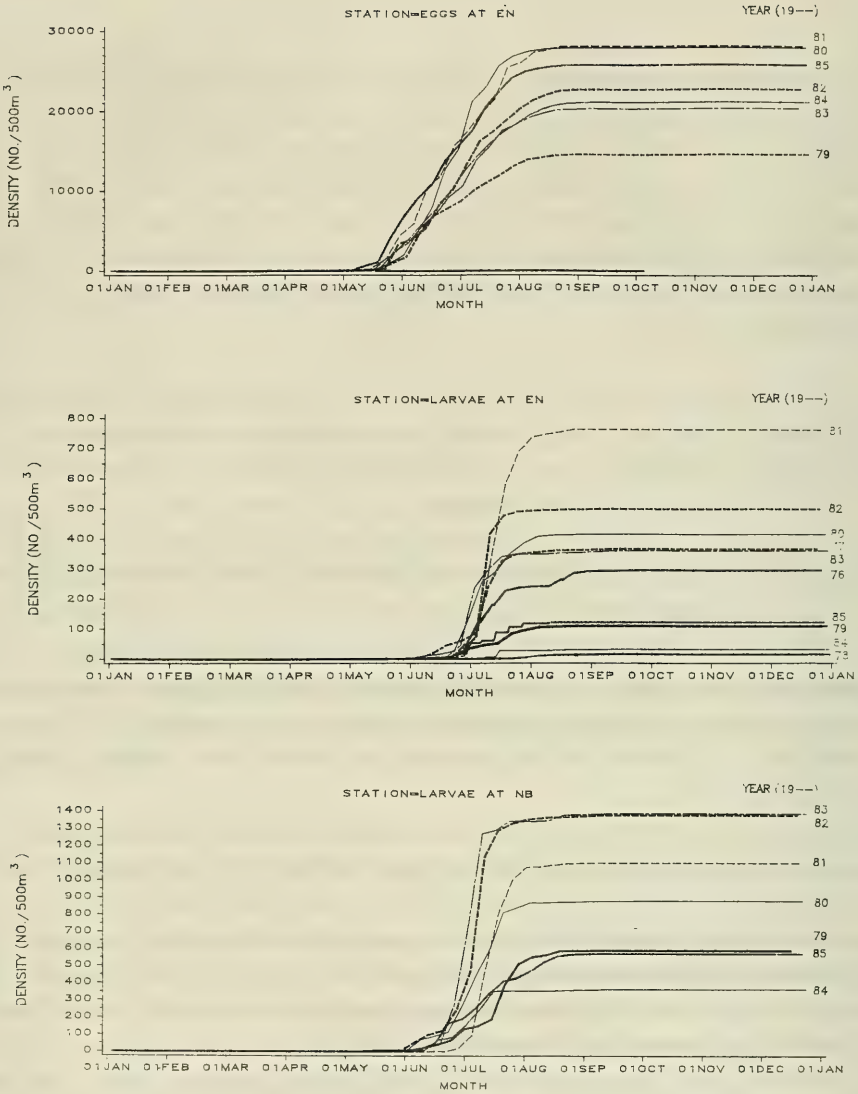


Figure 22. Annual cumulative density (no./500 m^3) of tautog eggs at EN and larvae at EN and NB.

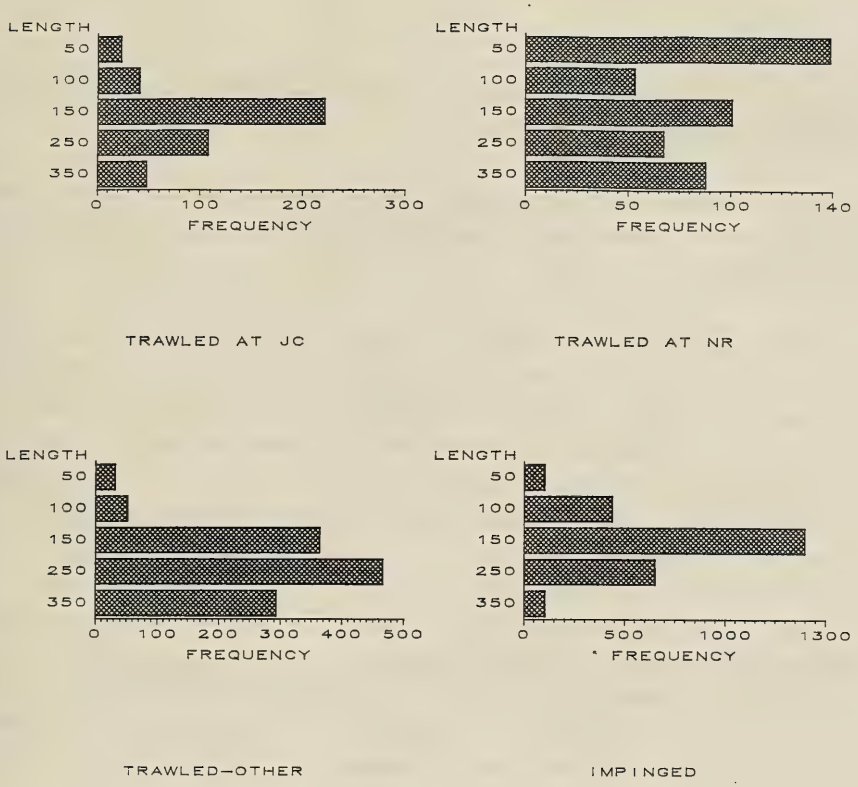


Figure 23. Tautog length frequency.

Tautogolabrus adspersus, cunner

The cunner (*Tautogolabrus adspersus*) is a coastal marine fish that prefers reef habitats (Bigelow and Schroeder 1953; Serchuk 1972; Olla et al. 1975, 1979; Dew 1976; Pottle and Green 1975). It ranges from northern Newfoundland to the mouth of the Chesapeake Bay (Leim and Scott 1966). Most cunner have limited home ranges and are active only during the day (Green 1975). Activity declines in cold weather and individuals become dormant at temperatures below 8°C and lie torpid among and under rocks (Green and Farwell 1971; Dew 1976; Olla et al. 1979). The cunner becomes mature in its first year (Dew 1976) and spawns inshore from May through August (Wheatland 1956). Eggs are pelagic and usually found in

the upper 5 m of the water column (Williams 1967). Metamorphosis of larvae is complete by 10 mm and juveniles move to the bottom (Miller 1958).

Cunner was found in all NUSCo programs, but it was most abundant in the plankton collections (Tables 10, 12 and 14). It ranked among the top ten fishes in all collections except seines (Appendices V, VIII, IX, X and XII).

Cunner eggs were the most abundant egg taxon entrained (Table 8) during the two-unit operational period. Annual egg entrainment estimates ranged from $1,675x10^6$ (1981) to $2,589x10^6$ (1983) (Table 9). Cunner egg density was lowest in either 1981 or 1982 depending on the abundance index used (median or mean). The lowest median was 2,958/500 m^3 in 1981 and the lowest mean was 1,761/500 m^3 in 1982. Similarly, egg density was highest in either 1983 (median = 5,934/500 m^3) or 1985 (mean = 3,016/500 m^3). Because the pattern of cunner abundance changed depending on the measure used, these data were not useful for describing trends.

Cunner was the fourth most abundant larval fish at NB and fifth at EN. Abundance at NB was higher than at EN in terms of both annual cumulative density (Fig. 24) and mean density (Tables 10 and 12). Further, temporal patterns at the two stations were different. Mean annual larval density was highest at EN (13.8/500 m^3) in 1981 and at NB (42.4/500 m^3) in 1983. Both indices were lower in 1984 and 1985 than in any year since 1979, but no trend was apparent.

Juvenile and adult cunner were present primarily from May through November in impingement and trawl collections; 25% of the total impingement catch occurred in June alone. Like tautog, cunner prefer the rocky habitats that surround MNPS. Unlike tautog, however, cunner contributed more than 1% to the total estimated impingment (Table 8). Annual impingement estimates varied an order of magnitude, from 466 in 1985 to 3,851 in 1982. As it was the case for tautog, the low number of cunner impinged during 1985 may have been related to the shut-down of Unit 2. Over 75% of cunner taken by trawl were caught at IN and JC (Table 15). The spatial distribution of cunner was also similar to that of tautog because both species have similar preferences and behavior. Although, juvenile cunner (individuals smaller than 75 mm) stay near shore and thus were more likely to be taken by trawls at both NR and JC (Fig. 25), juvenile tautog were abundant only at NR (Fig. 23). Smaller adults also stay inshore and were likely to be found in impingement collections; larger adults move offshore and were taken in trawls from deep

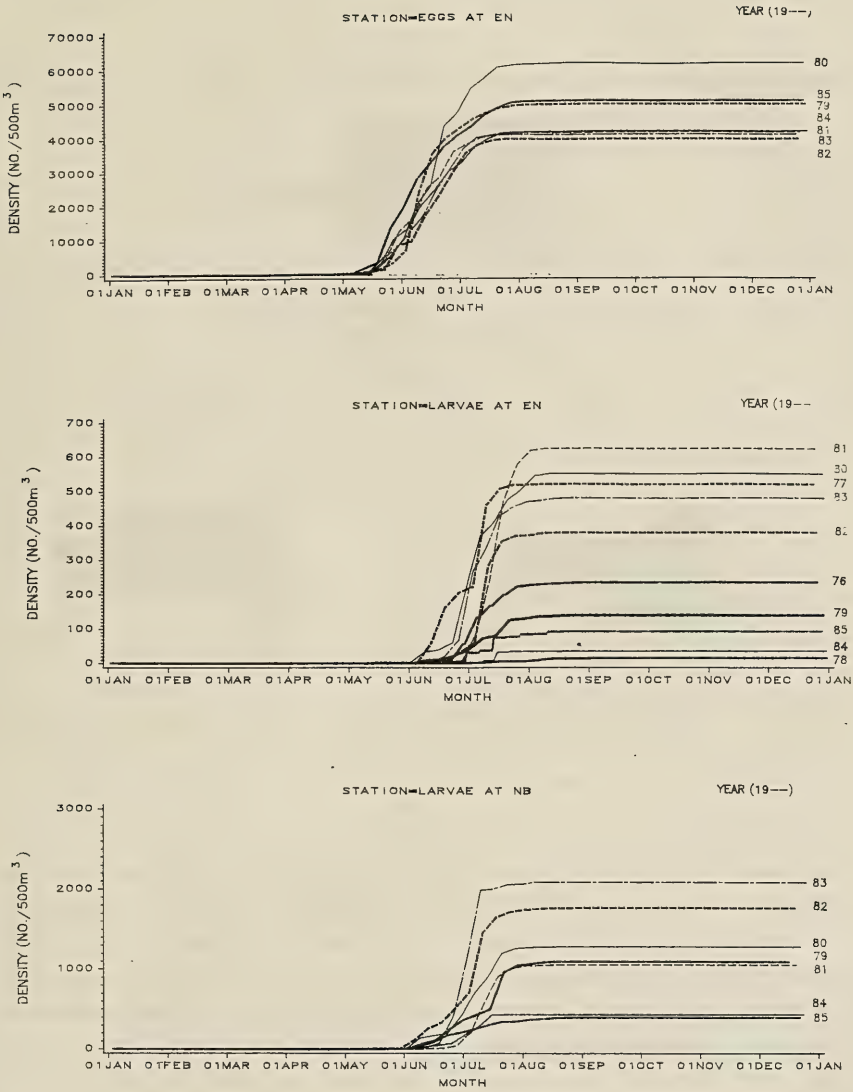


Figure 24. Annual cumulative density (no./500 m^3) of cunner eggs at EN and larvae at EN and NB.

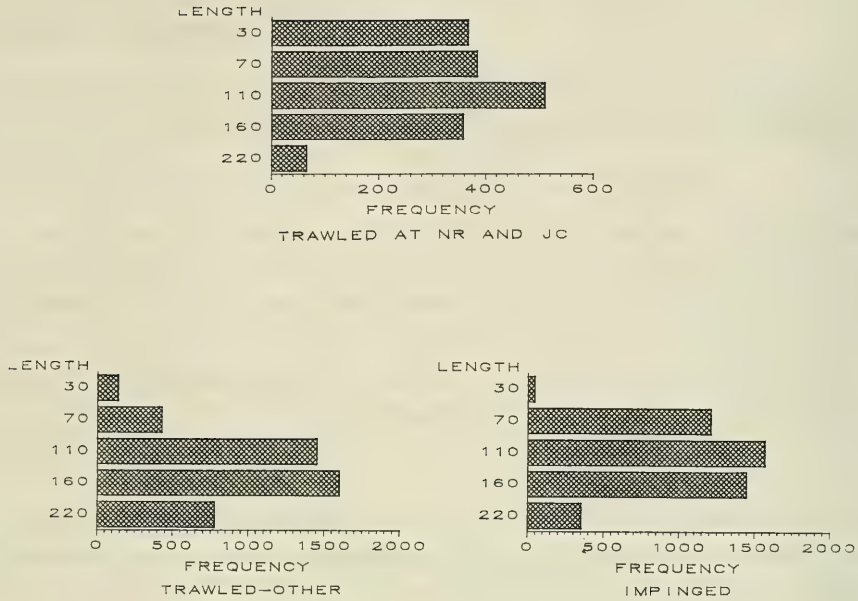


Figure 25. Length frequency of cunner.

water stations (Fig. 25). Although both total trawl catches (Table 14) and annual mean trawl catches (Fig. 26) have declined since 1979, the other indices of cunner abundance (e.g., egg and larval densities) did not follow the same trend. The precipitous drop of cunner larval abundance in 1984 could be explained by the predation that also affected the abundances of tautog and anchovies that year. A previous drop in cunner larval abundance in 1978, was followed in 1979 by the highest mean trawl catch (Fig. 26). None of these fluctutations could be attributed to MNPS operations.

The time-series models described the seasonal variability of cunner catches reasonably well as indicated by R^2 values of 0.70 for plankton and 0.65 for impingement. All the models included an annual cycle and the plankton models had an additional 4-mo cycle. The 1984 decline in larval abundance was well

described and is now a feature of the baseline model. Summaries of these models are presented in Appendices XXVI, XXVII, and XXVIII.

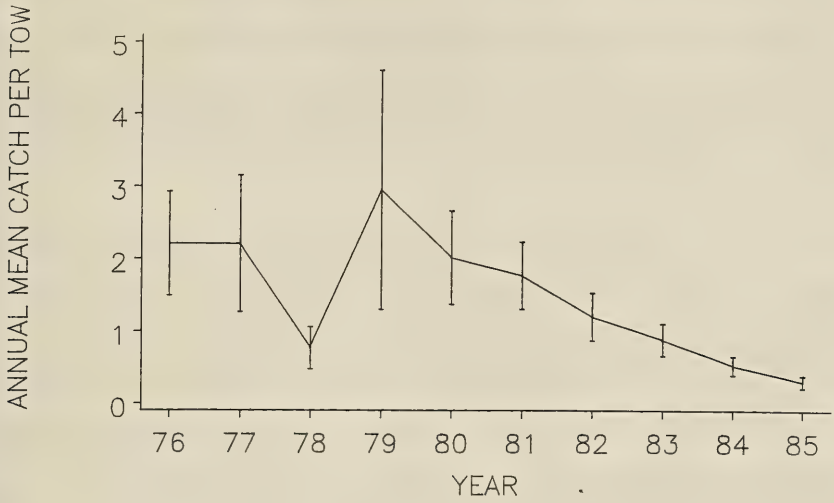


Figure 26. Annual mean catch of cunner taken by trawl; the vertical bars are approximated 95% confidence intervals for each year.

CONCLUSIONS

Impacts from the construction and operation of the Millstone Nuclear Power Station were assessed during the operation of Units 1 and 2, using representative collections of fish assemblages. The composition of these fish assemblages from January 1975 through December 1985 remained relatively stable, and was typical of that reported for LIS by other researchers. Fish abundances exhibited predictable seasonal and annual fluctuations; analyses indicated no adverse impact of two-unit operation. Data collected over the past 10 yr provide an excellent baseline for assessing the possible impacts from the operation of three power plants at MNPS.

SUMMARY

1. The construction and operation of the Millstone Nuclear Power Station (MNPS) could effect changes in fish assemblages in several ways. Larger fish may be removed from the population by impingement on the intake screens; eggs, larvae and small fish may be removed during entrainment through the cooling water system; and spatial distribution of local fish populations may change in response to the cooling water effluent.

2. Several programs were established to provide baseline data for assessing impacts of MNPS on fish assemblages: entrainment, offshore plankton, trawl, seine and impingement monitoring programs. These programs provided the data necessary for assessing the effects of two-unit operation and also provide the baseline for three-unit impact assessment.

3. Over 100 taxa of fish have been collected in the various Fish Ecology monitoring programs at MNPS from January 1976 through December 1985. Eight taxa were selected for detailed analysis based on their susceptibility to impact from impingement and entrainment: sand lance, anchovies, sticklebacks, silversides, tomcod, grubby, cunner and tautog.

4. The abundance of these taxa varied both seasonally and annually in all programs and to separate population fluctuations representing natural variability from those resulting from the construction and operation of MNPS, a time-series approach was developed and applied to the monitoring data.

5. The abundance of potentially impacted taxa remained relatively stable throughout the 10-yr period, except for larval and juvenile sand lance, and larval anchovy, cunner and tautog. Except for larval sand lance, these abundance changes were short-term. Large annual fluctuations of sand lance have been observed along the entire Atlantic coast. Thus the operation of two nuclear power plants at MNPS has not adversely affected fish abundance, distribution or species composition in the Millstone area of LIS.

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Appendix Ia. Summary of gill net sampling program. Letters indicate station sampled T = Twotree; I = Intake; J = Jordan Cove; C = Crescent Beach-Black Point; B = Bartlett Reef; N = Niantic Bay; E = East of effluent; W = West of effluent.

	J	F	M	A	M	J	J	A	S	O	N	D
1971	(NOTE: The following sampling program began using a 6-panel net)											T-I[a]
1972			T-I[a]			T-I[a]			T-I[a]			T-I[a]
1973			T-I[a]			T-I-J-C[a]			T-I J-C[a]			T-I J-C[a]
1974	T-I-J-C[a]											
1975	X[d]	T-I-J-C[a] B-N-E-W[b]			T-I-J-C[a]; B-N-E-W[b] NOTE: changed to an eight-panel net of varying mesh sizes starting in May.							
1976	X[d]	T-I-J-B-N-E-W[c]										
1977	T-I-J-B-N-E-W[c]								T-I-J-C-B-N-E-W[c]			
1978	T-I-J-C-B-N-E-W[c]											
1979 to 1982	T-I-J-C-B-N-E-W[c]											

[a] Indicates net was set at surface.

[b] Indicates net was set at bottom.

[c] Indicates a mid-water set.

[d] X = Same a December of the previous year.

Appendix Ib. Location of the gill net sampling sites.

Appendix II. Number of icthyoplankton samples from various stations during 1973-1985.

Entrainment stations	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
EN	189	288	692	934	906	866	942	936	776	774	338	338	262	8241
IN	472	726	333	.	.	.	.	.	.	.	.	.	.	1531
QCUT	189	288	132	.	.	.	.	.	.	.	.	.	.	609
Total	850	1302	1157	934	906	866	942	936	776	774	338	338	262	10381
Offshore stations	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
NB	79	124	155	21	20	28	209	238	236	238	142	128	119	1737
1	86	119	91	.	.	.	.	.	96	38	.		.	430
2	69	96	87	23	20	24	14	118	112	66	.	.	.	629
3	96	164	142	.	20	.	.	.	.	.	.	.	.	422
4	64	133	111	.	.	.	.	.	.	.	.	.	.	308
6	58	109	85	21	20	26	.	.	.	.	.	.	.	319
7	100	175	163	.	.	.	.	.	.	.	.	.	.	438
8	70	128	155	21	20	31	.	.	.	.	.	.	.	425
9	94	142	114	.	.	.	.	.	.	.	.	.	.	350
10	72	124	142	.	.	.	.	.	.	.	.	.	.	338
11	.	152	144	23	20	33	.	.	.	.	.	.	.	372
12	.	85	78	.	.	.	.	.	.	.	.	.	.	163
13	.	88	92	.	.	.	.	.	.	.	.	.	.	180
14		51	154	21	20	28	.	.	.	.	.	.	.	274
15	.	.	76	3	20	.	.	.	.	.	.	.	.	99
16	.	.	90	.	.	.	.	.	.	.	.	.	.	90
Total	788	1690	1879	133	160	170	223	356	444	342	142	128	119	6574

Appendix III. Number of offshore ichthyoplankton samples processed from NB for each month during 1973-1985.

Number of day samples

Month	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
1	.	.	15	1	1	1	2	4	4	4	4	2	2	40
2	.	10	10	1	1	1	2	4	6	8	6	2	2	53
3	.	8	12	1	1	1	4	6	4	8	4	2	3	54
4	.	10	8	1	1	5	16	16	16	12	10	10	10	115
5	2	8	15	3	3	4	20	20	16	16	9	8	8	132
6	12	10	12	4	3	4	16	20	16	20	9	8	8	142
7	8	10	11	3	3	4	20	16	20	16	8	10	8	137
8	13	13	6	3	3	4	16	16	20	20	10	8	10	142
9	8	11	8	1	1	1	3	4	4	4	4	4	2	55
10	10	10	3	1	1	1	2	6	4	4	2	6	3	53
11	9	8	3	1	1	1	2	4	4	4	3	2	2	44
12	4	12	3	1	1	1	2	4	4	4	2	2	2	42
Total	66	110	106	21	20	28	105	120	118	120	71	64	60	1009

Number of night samples

Month	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
1	.	.	1	.	.	.	2	4	4	4	4	2	2	23
2	.	1	5	.	.	.	2	4	6	6	6	2	2	34
3	.	1	7	.	.	.	4	4	4	8	4	2	2	36
4	.	1	3	.	.	.	20	16	16	12	10	9	9	96
5	.	1	5	.	.	.	16	20	16	16	9	9	9	101
6	1	1	5	.	.	.	16	20	16	20	9	8	8	104
7	1	3	8	.	.	.	20	16	20	16	8	10	8	110
8	5	1	3	.	.	.	16	16	20	20	10	8	10	109
9	1	1	3	.	.	.	2	4	4	4	4	4	2	29
10	1	2	3	.	.	.	2	6	4	4	3	6	3	34
11	4	1	3	.	.	.	2	4	4	4	2	2	2	28
12	.	1	3	.	.	.	2	4	4	4	2	2	2	24
Total	13	14	49	.	.	.	104	118	118	118	71	64	59	728

Appendix IV. Number of entrainment samples processed from each month during 1976-1985.

Larval samples

Month	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
1	77	84	78	84	81	72	72	34	36	10	628
2	72	72	71	72	75	72	72	32	34	13	585
3	84	80	34	78	78	78	84	38	34	33	621
4	78	75	57	75	78	81	78	32	34	36	624
5	78	78	84	84	78	75	72	34	37	36	656
6	78	80	78	75	78	81	84	37	33	24	648
7	78	75	77	78	81	81	78	33	36	28	645
8	78	83	79	84	75	78	78	38	36	26	655
9	81	78	75	72	78	78	78	34	32	24	630
10	75	78	77	84	84	30	24	9	10	14	485
11	78	68	81	78	60	21	30	9	8	8	441
12	77	54	75	78	90	29	24	8	8	10	453
Total	934	905	866	942	936	776	774	338	338	262	7071

Egg samples

Month	1979	1980	1981	1982	1983	1984	1985	Total
4		78	81	78	32	34	26	329
5	84	78	75	72	34	37	28	408
6	75	78	81	84	37	33	24	412
7	78	81	81	78	33	36	28	415
8	84	75	78	78	38	36	26	415
9	72	78	78	78	34	32	24	396
Total	393	468	474	468	208	208	156	2375

Appendix V. Percentage contributed by each taxa to the estimates of total entrainment and impingement at MNPS during 1976-1985.

Taxa	Entrainment		Impingement
	Larvae	Eggs	
Anchoa spp.	61.3	10.8	8.1
Pseudopleuronectes americanus	10.8	< 0.1	8.5
Ammodytes americanus	9.7	< 0.1	47.8
Myoxocephalus aenaeus	3.7	< 0.1	5.9
Tautogolabrus adspersus	1.8	55.5	1.9
Pholis gunnellus	1.8	0.0	0.1
Brevoortia tyrannus	1.8	< 0.1	0.2
Tautoga onitis	1.8	27.2	0.8
Syngnathus fuscus	1.0	0.0	1.6
Liparis spp.	1.0	0.0	0.1
Ulvaria subbifurcata	0.9	0.0	< 0.1
Scophthalmus aquosus	0.7	1.4	0.9
Peprilus triacanthus	0.7	< 0.1	1.5
Enchelyopus cimbrius	0.6	0.4	< 0.1
Prionotus spp.	0.3	2.2	0.2
Gobiidae	0.3	0.0	0.0
Stenotomus chrysops	0.2	1.0	0.1
Myoxocephalus octodecemspinosus	0.2	0.0	< 0.1
Cynoscion regalis	0.2	< 0.1	0.2
Scomber scombrus	0.2	< 0.1	< 0.1
Anguilla rostrata	0.2	0.0	0.2
Paralichthys oblongus	0.2	< 0.1	< 0.1
Menidia spp.	0.1	0.1	5.5
Clupea harengus	0.1	< 0.1	< 0.1
Clupeidae	0.1	0.0	< 0.1
Urophycis spp.	0.1	0.3	0.1
Sphoeroides maculatus	0.1	< 0.1	0.2
Gadus morhua	0.1	< 0.1	< 0.1
Merluccius bilinearis	0.1	0.0	1.6
Microgadus tomcod	0.1	0.0	3.4
Paralichthys dentatus	< 0.1	< 0.1	0.2
Sciaenidae	< 0.1	0.0	0.0
Trinectes maculatus	< 0.1	< 0.1	0.1
Etropus microstomus	< 0.1	0.0	< 0.1
Gasterosteus aculeatus	< 0.1	0.0	5.1
Centropristis striata	< 0.1	0.0	< 0.1
Ophidion marginatum	< 0.1	0.0	< 0.1
Alosa spp.	< 0.1	1.0	< 0.1
Mentlcirrhus saxatilis	< 0.1	< 0.1	< 0.1
Lophius americanus	< 0.1	0.0	< 0.1
Osmerus mordax	< 0.1	0.0	0.4
Apeltes quadracus	< 0.1	0.0	< 0.1
Gadidae	< 0.1	< 0.1	< 0.1
Limanda ferruginea	< 0.1	0.0	0.0
Pollachius virens	< 0.1	0.0	0.2
Bothidae	< 0.1	0.0	0.0
Chaetodon ocellatus	< 0.1	0.0	< 0.1
Lumpenus lumpretaeformis	< 0.1	0.0	0.0
Cyclopterus lumpus	< 0.1	0.0	0.5
Gasterosteus wheatlandi	< 0.1	0.0	1.7
Mugil cephalus	< 0.1	0.0	< 0.1
Hippocampus erectus	< 0.1	0.0	< 0.1

Appendix V. (continued)

Taxa	Entrainment		Impingement
	Larvae	Eggs	
Pungitius pungitius	< 0.1	0.0	< 0.1
Fundulus spp.	< 0.1	< 0.1	0.1
Conger oceanicus	< 0.1	0.0	< 0.1
Hemitripterus americanus	< 0.1	< 0.1	0.1
Alectis ciliaris	0.0	0.0	< 0.1
Alosa aestivalis	0.0	0.0	0.5
Alosa mediocris	0.0	0.0	< 0.1
Alosa pseudoharengus	0.0	0.0	0.2
Alosa sapidissima	0.0	0.0	< 0.1
Aluterus schoepfi	0.0	0.0	< 0.1
Aulostomus maculatus	0.0	0.0	< 0.1
Bairdiella chrysoura	0.0	0.0	< 0.1
Brosme brosme	0.0	0.0	< 0.1
Caranx crysos	0.0	0.0	< 0.1
Caranx hippos	0.0	0.0	< 0.1
Chilomycterus schoepfi	0.0	0.0	< 0.1
Cyprinodon variegatus	0.0	< 0.1	< 0.1
Dactylopterus volitans	0.0	0.0	< 0.1
Decapterus macarellus	0.0	0.0	< 0.1
Decapterus punctatus	0.0	0.0	< 0.1
Etrumeus teres	0.0	0.0	< 0.1
Fistularia tabacaria	0.0	0.0	< 0.1
Hippocampus spp.	0.0	0.0	< 0.1
Ictalurus catus	0.0	0.0	< 0.1
Leiostomus xanthurus	0.0	0.0	< 0.1
Lepomis macrochirus	0.0	0.0	< 0.1
Macrozoarces americanus	0.0	0.0	< 0.1
Melanogrammus aeglefinus	0.0	0.0	< 0.1
Monacanthus hispidus	0.0	0.0	< 0.1
Monocanthus spp.	0.0	0.0	< 0.1
Morone americana	0.0	< 0.1	0.7
Morone saxatilis	0.0	0.0	< 0.1
Mugil curema	0.0	0.0	< 0.1
Mustelis canis	0.0	0.0	< 0.1
Myoxocephalus spp.	0.0	0.0	< 0.1
Ophidiidae	0.0	0.0	< 0.1
Ophidion welshi	0.0	0.0	< 0.1
Opsanus tau	0.0	0.0	0.2
Petromyon marinus	0.0	0.0	< 0.1
Pomatomus saltatrix	0.0	0.0	0.1
Priacanthus arenatus	0.0	0.0	< 0.1
Priacanthus cruentatus	0.0	0.0	< 0.1
Pristigenys alta	0.0	0.0	< 0.1
Raja spp.	0.0	0.0	0.5
Rhinoptera bonasus	0.0	0.0	< 0.1
Salmo trutta	0.0	0.0	< 0.1
Selar crumenopthalmus	0.0	0.0	< 0.1
Selene setapinnis	0.0	0.0	< 0.1
Selene vomer	0.0	0.0	< 0.1
Seriola zonata	0.0	0.0	< 0.1
Sphyraena borealis	0.0	0.0	< 0.1
Squalus acanthias	0.0	0.0	< 0.1
Trachurus lathami	0.0	0.0	< 0.1

Appendix VI. Annual mean fish larval density (no./500m^3) at EN during 1976-1985.

Taxa	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Mean
Anchoa spp.	185.2	178.2	72.0	313.3	346.7	550.5	182.3	301.7	66.3	245.2	244.1
Pseudopleuronectes americanus	42.0	27.6	31.0	26.8	59.5	31.4	59.2	70.6	45.9	36.2	43.0
Ammodytes americanus	10.8	55.2	110.8	50.6	65.7	52.8	12.4	13.4	8.3	4.7	38.5
Myoxocephalus aenaeus	4.7	11.7	7.1	10.7	11.2	25.5	20.8	20.5	16.0	19.9	14.8
Tautogolabrus adspersus	5.6	12.1	0.4	3.2	12.4	13.8	8.6	12.5	1.0	2.8	7.2
Pholis gunnellus	2.4	5.1	4.3	6.3	5.5	18.5	10.9	5.1	5.9	7.3	7.1
Brevoortia tyrannus	2.9	1.8	2.1	0.4	1.9	1.6	7.5	19.9	1.9	30.6	7.1
Tautoga onitis	7.0	8.5	0.4	2.5	9.3	16.7	11.1	9.5	0.8	4.5	7.0
Syngnathus fuscus	1.5	3.5	2.2	8.5	3.4	6.1	3.2	4.7	4.0	2.7	4.0
Liparis spp.	1.1	6.1	5.9	2.4	4.2	4.7	1.3	3.2	1.9	7.7	3.8
Ulvaria subbifurcata	3.5	1.3	1.9	2.9	3.6	3.8	4.3	1.9	1.0	11.8	3.6
Scophthalmus aquosus	4.0	6.0	0.7	2.2	2.5	2.9	1.5	7.0	1.7	0.6	2.9
Peprilus triacanthus	4.7	1.1	0.4	0.5	9.9	4.7	3.1	3.0	0.4	0.7	2.8
Enchelyopus cimbrius	1.2	2.5	1.8	2.1	1.9	0.5	2.3	4.7	1.7	3.0	2.2
Prionotus spp.	0.8	0.6	0.1	0.3	0.9	5.2	0.3	1.6	0.2	2.0	1.2
Gobiidae	2.9	2.0	0.7	0.2	0.7	0.1	0.3	0.7	2.2	1.8	1.2
Stenotomus chrysops	1.1	1.8	0.1	1.0	1.1	2.0	0.4	0.0	< 0.1	< 0.1	0.7
Myoxocephalus octodecemspinosus	0.2	0.3	0.2	0.3	0.3	0.5	1.1	2.2	1.5	0.7	0.7
Cynoscion regalis	0.2	0.7	< 0.1	1.2	< 0.1	1.5	1.6	0.8	0.1	0.5	0.7
Scomber scombrus	0.6	3.7	0.1	0.1	0.1	< 0.1	< 0.1	1.3	< 0.1	0.3	0.6
Anguilla rostrata	0.4	1.9	1.1	1.0	0.2	0.3	0.3	0.4	0.2	0.1	0.6
Paralichthys oblongus	0.6	0.8	0.1	0.1	0.1	0.8	1.3	0.4	0.4	1.0	0.6
Menidia spp.	0.5	0.1	0.5	0.6	0.5	0.4	0.2	2.2	0.1	0.3	0.5
Clupea harengus	0.1	0.3	0.2	0.4	< 0.1	2.0	0.3	0.0	0.2	0.3	0.4
Clupeidae	< 0.1	0.6	0.3	0.4	0.2	0.1	< 0.1	0.4	0.1	0.0	0.2
Urophycis spp.	0.9	0.1	< 0.1	< 0.1	< 0.1	0.8	< 0.1	0.1	< 0.1	0.2	0.2
Sphoeroides maculatus	0.6	0.2	0.0	< 0.1	< 0.1	0.1	0.2	0.2	< 0.1	0.5	0.2
Gadus morhua	0.1	< 0.1	< 0.1	< 0.1	0.1	1.1	0.2	0.2	< 0.1	< 0.1	0.2
Paralichthys dentatus	0.1	< 0.1	0.1	0.1	0.2	0.1	0.1	0.3	< 0.1	0.3	0.1
Merluccius bilinearis	0.4	0.2	< 0.1	0.1	0.2	0.1	< 0.1	0.1	0.1	0.1	0.1
Microgadus tomcod	0.1	0.1	0.1	0.1	< 0.1	0.1	0.3	0.2	0.1	0.1	0.1
Sciaenidae	< 0.1	0.0	< 0.1	0.0	< 0.1	1.0	0.0	< 0.1	< 0.1	0.1	0.1
Trinectes maculatus	0.1	0.2	0.0	< 0.1	< 0.1	0.6	< 0.1	0.1	< 0.1	0.0	0.1
Etropus microstomus	0.3	0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	0.1	0.1	< 0.1	0.1
Gasterosteus aculeatus	0.1	0.1	0.2	0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.0	0.0	0.0
Centropristis striata	0.1	0.1	< 0.1	< 0.1	0.1	0.1	0.0	0.0	< 0.1	0.0	0.0
Ophidion marginatum	0.2	< 0.1	0.0	0.0	< 0.1	0.0	0.0	0.0	0.1	< 0.1	0.0
Alosa spp.	0.0	0.0	< 0.1	0.1	< 0.1	< 0.1	0.0	0.1	0.1	0.0	0.0
Menticirrhus saxatilis	< 0.1	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0
Lophius americanus	0.0	0.0	0.0	< 0.1	0.2	< 0.1	0.0	0.0	< 0.1	0.0	0.0
Osmerus mordax	< 0.1	0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.0	0.0	0.0	0.0
Apeltes quadracus	< 0.1	< 0.1	< 0.1	0.0	< 0.1	< 0.1	0.0	< 0.1	0.0	0.2	0.0
Gadidae	< 0.1	0.1	< 0.1	< 0.1	< 0.1	0.1	0.0	< 0.1	< 0.1	< 0.1	0.0
Limanda ferruginea	0.2	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.0
Pollachius virens	< 0.1	< 0.1	< 0.1	0.0	< 0.1	< 0.1	< 0.1	0.1	< 0.1	< 0.1	0.0
Bothidae	0.0	0.0	0.0	0.0	0.0	0.0	< 0.1	0.0	0.0	0.0	< 0.1
Chaetodon ocellatus	0.0	0.0	0.0	0.0	< 0.1	0.0	0.0	0.0	0.0	0.0	< 0.1
Lumpenus lumpretaeformis	0.0	0.0	0.0	< 0.1	0.0	0.0	0.0	0.0	0.0	0.0	< 0.1
Cyclopterus lumpus	0.0	0.0	0.0	0.0	0.0	< 0.1	0.0	< 0.1	0.0	0.0	< 0.1
Mugil cephalus	< 0.1	< 0.1	0.0	0.0	0.0	< 0.1	0.0	0.0	0.0	0.0	< 0.1
Hippocampus erectus	< 0.1	0.0	0.0	0.0	< 0.1	0.0	< 0.1	0.0	0.0	< 0.1	< 0.1
Pungitius pungitius	0.0	0.0	< 0.1	< 0.1	0.0	< 0.1	< 0.1	0.0	0.0	0.0	< 0.1
Fundulus spp.	< 0.1	< 0.1	< 0.1	< 0.1	0.0	< 0.1	< 0.1	< 0.1	0.0	0.0	< 0.1
Conger oceanicus	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.0	< 0.1	< 0.1	< 0.1
Hemitripterus americanus	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Total	288.4	335.9	246.5	440.0	544.6	752.2	336.8	490.0	163.8	387.2	398.5

Appendix VII. Annual mean fish larval density (no./500^3) at NB during 1979-1985.

Taxa	1979	1980	1981	1982	1983	1984	1985	Mean
Anchoa spp.	320.7	418.2	494.1	292.5	485.3	62.9	239.3	330.4
Ammodytes americanus	56.2	52.4	174.9	9.4	40.9	27.9	11.4	53.3
Pseudopleuronectes americanus	19.0	42.9	26.8	43.1	68.4	41.8	44.2	40.9
Tautogolabrus adspersus	19.6	24.9	17.6	34.4	42.4	7.9	8.0	22.1
Tautoga onitis	10.6	17.3	18.4	27.8	28.3	6.7	11.3	17.2
Myoxocephalus aenaeus	6.1	9.1	13.0	11.2	11.8	9.6	11.9	10.4
Brevoortia tyrannus	0.6	1.9	2.4	36.1	9.8	9.6	6.9	9.6
Scophthalmus aquosus	8.3	5.8	3.0	10.6	12.2	5.7	3.6	7.0
Pholis gunnellus	4.6	3.6	12.1	6.6	3.8	6.4	4.5	5.9
Peprilus triacanthus	1.9	5.3	6.9	10.4	11.5	0.8	4.7	5.9
Enchelyopus cimbrius	6.2	2.6	1.1	6.8	6.3	8.8	5.5	5.3
Syngnathus fuscus	2.7	3.2	3.7	2.4	6.5	3.6	1.5	3.4
Prionotus spp.	1.3	1.5	8.4	5.1	4.5	0.9	1.7	3.3
Liparis spp.	1.8	2.3	3.9	0.9	3.0	0.8	6.3	2.7
Ulvaria subbifurcata	1.3	2.2	2.2	4.8	1.9	1.4	2.9	2.4
Myoxocephalus octodecemspinosus	0.5	1.3	2.3	2.0	3.1	3.9	1.6	2.1
Paralichthys oblongus	0.2	0.6	1.7	1.9	3.0	0.2	1.6	1.3
Cynoscion regalis	1.6	0.1	2.3	3.7	0.1	0.0	0.3	1.2
Stenotomus chrysops	2.1	4.0	1.4	0.4	0.1	< 0.1	< 0.1	0.9
Scomber scombrus	1.6	0.3	< 0.1	0.9	1.0	0.5	1.0	0.7
Merluccius bilinearis	0.5	0.5	0.4	0.4	0.2	0.4	0.7	0.4
Paralichthys dentatus	0.2	0.2	1.2	0.1	0.2	0.3	0.2	0.3
Clupeidae	0.1	0.1	0.1	0.0	< 0.1	< 0.1	3.4	0.3
Gobiidae	0.2	0.0	0.1	0.2	0.5	0.4	0.3	0.2
Sciaenidae	< 0.1	0.0	1.2	0.4	1.1	< 0.1	0.1	0.2
Clupea harengus	0.1	0.0	1.0	0.2	0.1	< 0.1	0.6	0.2
Etropus microstomus	0.0	0.0	0.1	0.2	0.3	0.1	0.2	0.1
Gadus morhua	< 0.1	< 0.1	1.3	0.1	0.5	0.3	0.1	0.1
Menidia spp.	0.4	0.2	0.6	0.0	0.6	< 0.1	< 0.1	0.1
Alosa spp.	0.1	< 0.1	0.2	0.0	0.4	0.1	0.2	< 0.1
Pomatomus saltatrix	0.0	0.0	0.0	0.0	0.0	0.0	0.1	< 0.1
Sphoeroides maculatus	0.2	0.1	0.1	0.1	< 0.1	0.1	0.1	< 0.1
Anguilla rostrata	0.1	0.1	0.1	0.1	< 0.1	0.1	0.1	< 0.1
Centropristis striata	< 0.1	< 0.1	0.2	0.0	1.0	0.0	0.0	< 0.1
Urophycis spp.	0.1	< 0.1	0.6	0.2	< 0.1	0.1	0.2	< 0.1
Pollachius virens	0.0	0.0	0.0	0.0	0.1	0.1	< 0.1	< 0.1
Cyclopterus lumpus	0.0	0.0	< 0.1	0.0	0.0	0.0	0.0	< 0.1
Fundulus spp.	0.0	0.0	< 0.1	0.0	0.0	0.0	0.0	< 0.1
Hippocampus erectus	0.0	0.0	0.0	0.0	0.0	0.0	< 0.1	< 0.1
Microgadus tomcod	< 0.1	0.1	0.2	0.1	0.1	< 0.1	0.1	< 0.1
Gadidae	0.0	0.1	0.0	< 0.1	< 0.1	0.1	0.0	< 0.1
Conger oceanicus	< 0.1	0.0	0.0	0.0	< 0.1	0.1	0.0	< 0.1
Lophius americanus	< 0.1	0.0	0.0	0.1	< 0.1	0.0	0.0	< 0.1
Gasterosteus aculeatus	< 0.1	0.0	< 0.1	0.0	0.0	0.0	0.0	< 0.1
Osmerus mordax	< 0.1	0.0	0.0	< 0.1	0.0	0.0	0.0	< 0.1
Ophidion marginatum	0.1	< 0.1	< 0.1	0.0	0.1	< 0.1	0.4	< 0.1
Trinectes maculatus	0.0	0.0	0.4	< 0.1	< 0.1	0.0	< 0.1	< 0.1
Hemitripterus americanus	0.0	< 0.1	< 0.1	0.1	< 0.1	0.0	0.0	< 0.1
Limanda ferruginea	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.2	0.1	< 0.1
Total	469.1	601.1	804.0	513.1	749.4	201.9	375.2	530.5

Appendix VIII. Annual mean fish egg density (no./500m^3) at EN during April through September of 1979-1985.

Taxa	1979	1980	1981	1982	1983	1984	1985	Mean
Tautogolabrus adspersus	2454.5	2898.5	1908.9	1761.6	2001.2	2240.7	3016.3	2326.0
Tautoga onitis	714.6	1311.4	1273.9	1033.5	999.2	1122.0	1507.5	1137.4
Anchoa spp.	413.0	306.1	339.4	218.4	636.7	1196.9	53.2	452.0
Prionotus spp.	30.4	75.6	115.5	118.3	133.2	51.9	112.2	91.0
Scophthalmus aquosus	34.5	41.4	53.4	24.6	17.3	41.9	208.9	60.3
Alosa spp.	267.3	0.3	0.2	14.6	0.0	0.0	0.0	40.3
Stenotomus chrysops	10.6	0.3	63.6	53.3	45.9	95.4	11.6	40.1
Enchelyopus cimbrius	14.2	9.5	17.4	31.6	7.7	8.9	16.3	15.1
Urophycis spp.	2.0	6.5	23.5	1.7	14.3	30.8	21.7	14.4
Menidia spp.	0.0	3.4	1.5	0.3	0.7	19.8	< 0.1	3.7
Brevoortia tyrannus	0.8	< 0.1	0.5	0.6	2.0	6.4	3.8	2.0
Cynoscion regalis	4.9	3.0	1.3	0.0	0.0	0.0	0.0	1.3
Scomber scombrus	5.2	0.5	0.0	2.2	0.0	0.0	0.0	1.1
Pseudopleuronectes americanus	0.8	0.9	4.7	1.1	0.3	0.0	0.0	1.1
Trinectes maculatus	5.8	0.0	0.1	0.0	0.0	0.0	0.0	0.8
Hemitripterus americanus	0.0	0.0	0.0	4.9	0.0	0.0	0.0	0.7
Myoxocephalus aenaeus	3.2	< 0.1	< 0.1	0.9	< 0.1	0.0	0.5	0.6
Peprilus triacanthus	0.7	1.1	0.0	0.0	0.0	0.0	0.0	0.3
Paralichthys oblongus	0.2	0.5	0.0	0.5	0.0	0.0	0.0	0.2
Cyprinodon variegatus	< 0.1	0.0	0.0	0.7	0.0	0.0	0.0	0.1
Paralichthys dentatus	0.0	0.1	0.0	0.3	0.0	0.0	0.0	0.1
Ammodytes americanus	0.3	0.0	0.0	0.0	0.0	0.0	0.0	< 0.1
Fundulus spp.	0.3	0.0	0.0	< 0.1	0.0	0.0	0.0	< 0.1
Morone americana	0.0	0.0	0.0	< 0.1	0.0	0.2	0.0	< 0.1
Myoxocephalus octodecemspinosus	0.1	0.0	0.0	0.0	0.0	0.0	0.0	< 0.1
Clupea harengus	< 0.1	0.0	0.0	0.0	0.0	0.0	0.0	< 0.1
Sphoeroides maculatus	< 0.1	0.0	0.0	0.0	0.0	0.0	0.0	< 0.1
Gadidae	0.0	< 0.1	< 0.1	< 0.1	< 0.1	0.0	< 0.1	< 0.1
Gadus morhua	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.0	0.0	< 0.1
Total	3963.6	4659.1	3803.9	3269.1	3858.7	4814.9	4952.2	4188.8

Appendix IX. Annual impingement estimate for all taxa impinged, calculated using flows from 0800-0745 (Units 1 & 2 combined by year except during 1984-85 when Unit 2 was estimated alone).

Fish Taxa	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
Ammodytes spp.	65	69	277	98	192	269	136	449	485411	73	487039
Pseudopleuronectes americanus	5654	7622	7676	23544	7207	7640	8875	13467	2542	2765	86992
Anchoa spp.	5606	804	869	3340	4426	4755	5895	52280	4200	342	82517
Myoxocephalus aenaeus	2108	2357	7528	3699	10736	5450	6486	14634	2359	4553	59866
Menidia spp.	1585	1328	12155	12187	10199	3733	3872	8136	1042	1480	55717
Microgadus tomcod	91	339	2398	1455	1314	8121	11868	2860	4938	1129	34513
Gasterosteus spp.	2411	5375	5511	9918	7441	.	.	.	.	.	30656
Gasterosteus aculeatus	.	.	.	.	.	6817	2951	9472	1055	852	21147
Tautogolabrus adspersus	903	1429	1862	3110	1157	2566	3851	2900	1188	466	19432
Gasterosteus wheatlandi	.	.	.	.	.	601	1393	14381	702	21	17098
Syngnathus fuscus	643	384	1265	1289	1152	1611	1029	6572	1467	456	15868
Merluccius bilinearis	791	1086	545	837	1703	679	560	9419	133	105	15858
Peprilus triacanthus	135	149	298	1574	1139	1829	4061	3086	1455	1336	15062
Scophthalmus aquosus	679	454	406	1024	1122	640	743	3401	569	173	9211
Tautoga onitis	883	809	1074	866	338	814	1579	1512	664	122	8661
Morone americana	312	761	670	368	643	598	2540	912	375	48	7227
Cyclopterus lumpus	47	525	781	578	1301	329	11	859	120	1010	5561
Alosa aestivalis	121	221	207	381	162	280	125	3605	91	51	5244
Raja spp.	499	271	240	337	453	531	507	1468	275	98	4679
Osmerus mordax	399	337	658	476	175	438	492	1479	71	97	4622
Brevoortia tyrannus	200	159	104	187	222	165	306	682	167	242	2434
Paralichthys dentatus	463	127	87	14	80	390	349	241	646	29	2426
Alosa pseudoharengus	97	359	52	255	156	223	273	659	79	59	2212
Sphoeroides maculatus	289	153	34	61	49	130	469	712	174	86	2157
Prionotus spp.	503	226	96	89	87	323	255	223	49	72	1923
Cynoscion regalis	31	886	84	10	12	632	58	90	34	14	1851
Anguilla rostrata	99	210	76	180	115	114	486	379	60	48	1767
Pollachius virens	9	19	11	2	89	164	161	888	253	0	1596
Opsanus tau	163	103	167	169	155	228	214	246	98	28	1571
Liparis spp.	9	308	188	48	53	371	55	272	39	66	1409
Fundulus spp.	58	75	464	97	98	340	138	80	20	8	1378
Pomatomus saltatrix	53	133	14	206	55	192	65	459	110	46	1333
Stenotomus chrysops	153	118	30	139	48	101	213	343	53	105	1303
Hemitripterus americanus	12	20	11	35	56	132	240	365	22	6	899
Pholis gunnellus	72	39	78	111	67	119	100	247	49	12	894
Trinectes maculatus	28	15	30	119	55	61	139	88	194	21	750
Urophycis chuss	7	18	2	115	33	51	47	198	71	13	555
Urophycis regia	5	0	0	3	18	24	25	444	7	0	526
Clupea harengus	60	128	0	5	2	77	21	48	12	12	365
Caranx hippos	6	0	2	6	2	76	134	76	21	34	357
Morone saxatilis	4	10	3	11	0	4	235	67	0	7	341
Melanogrammus aeglefinus	0	0	6	307	7	0	0	0	0	0	320
Gadus morhua	0	0	0	0	0	10	65	74	142	7	298
Monacanthus hispidus	6	30	50	118	8	6	11	7	3	9	248

Appendix IX. (continued)

Fish Taxa	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
Etropus microstomus	3	5	0	0	6	5	9	178	20	10	236
Urophycis spp.	0	34	190	4	2	2	0	3	0	0	235
Mugil cephalus	8	13	23	11	37	37	9	34	39	4	215
Alosa sapidissima	12	1	0	5	33	33	25	82	16	0	207
Myoxocephalus spp.	1	203	0	0	0	0	0	0	0	0	204
Leiostomus xanthurus	16	4	152	0	0	0	2	28	0	0	202
Centropristis striata	31	6	0	0	9	2	12	54	74	0	188
Sphyraena borealis	12	4	12	0	0	35	71	2	12	0	148
Ophidion marginatum	26	0	0	10	0	0	9	31	50	6	132
Apeltes quadracus	2	5	8	10	33	48	14	5	2	0	127
Paralichthys oblongus	50	15	6	14	6	16	2	7	0	6	122
Urophycis tenuis	0	2	11	0	6	17	30	6	45	0	117
Scomber scombrus	3	4	5	17	9	12	48	9	0	0	107
Selene setapinnis	30	0	0	0	0	0	2	6	20	34	92
Aluterus schoepfi	1	57	6	0	3	6	9	0	6	0	88
Selene vomer	0	37	4	0	5	0	2	4	14	22	88
Mustelis canis	2	11	34	12	0	6	7	5	0	8	85
Ulvaria subbifurcata	6	5	9	13	0	7	7	26	6	4	83
Conger oceanicus	17	8	0	0	2	2	9	31	0	0	69
Caranx crysos	0	13	7	0	0	5	14	4	24	0	67
Mugil curema	0	0	0	0	0	24	0	7	27	8	66
Pungitius pungitius	0	0	2	0	0	6	21	22	5	10	66
Alectis ciliaris	1	7	2	0	3	30	2	0	6	7	58
Chaetodon ocellatus	0	0	4	0	0	4	12	26	0	7	53
Myoxocephalus octodecemspinosus	0	0	3	5	5	14	7	10	0	0	44
Squalus acanthias	2	0	0	3	6	2	0	31	0	0	44
Trachurus lathami	0	0	0	32	0	4	0	4	0	0	40
Ophidiidae	31	0	0	0	0	0	5	0	0	0	36
Fistularia tabacaria	3	4	5	0	10	2	5	0	0	6	35
Cyprinodon variegatus	0	0	13	0	4	2	5	9	0	0	33
Bairdiella chrysoura	0	0	0	0	0	0	2	24	0	0	26
Dactylopterus volitans	0	2	0	0	0	2	10	3	0	7	24
Etrumeus teres	0	2	0	19	0	0	2	0	0	0	23
Gadidae	0	0	5	0	2	0	13	0	0	0	20
Hippocampus erectus	2	0	0	0	0	2	0	11	0	3	18
Lophius americanus	0	0	0	0	2	0	9	0	0	7	18
Chilomycterus schoepfi	0	2	3	0	0	2	2	0	8	0	17
Decapterus macarellus	0	0	0	0	0	0	0	0	15	0	15
Pristigenys alta	0	0	0	0	0	0	2	0	12	0	14
Decapterus punctatus	0	0	0	0	0	4	7	2	0	0	13
Menticirrhus saxatilis	0	0	4	5	0	0	2	2	0	0	13
Alosa spp.	0	0	0	0	0	0	5	0	6	0	11
Priacanthus arenatus	0	0	0	0	0	0	5	0	6	0	11

Appendix IX. (continued)

Fish Taxa	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
Priacanthus cruentatus	0	0	0	0	0	2	0	9	0	0	11
Macrozoarces americanus	0	0	0	2	0	0	0	2	0	6	10
Seriola zonata	0	9	0	0	0	0	0	0	0	0	9
Enchelyopus cimbrius	0	0	0	0	0	0	0	0	8	0	8
Monocanthus spp.	0	0	0	0	0	0	2	3	3	0	8
Clupeidae	0	0	0	0	0	0	0	0	3	4	7
Hippocampus spp.	0	0	0	0	0	0	0	0	0	6	6
Salmo trutta	1	0	0	5	0	0	0	0	0	0	6
Selar crumenopthalmus	3	0	0	0	0	2	0	0	0	0	5
Alosa mediocris	0	0	0	0	0	0	2	2	0	0	4
Ictalurus catus	2	0	0	0	0	0	2	0	0	0	4
Ophidion welshi	0	0	0	0	0	2	0	2	0	0	4
Aulostomus maculatus	3	0	0	0	0	0	0	0	0	0	3
Brosme brosme	1	0	0	0	0	0	0	2	0	0	3
Lepomis macrochirus	0	0	0	0	0	0	0	2	0	0	2
Petromyon marinus	0	0	0	0	2	0	0	0	0	0	2
Rhinoptera bonasus	0	0	0	0	0	2	0	0	0	0	2
Total	25528	27909	46517	67535	52512	51973	61436	158468	511387	16266	1019531

Invertebrate Taxa	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
Loligo pealei	13836	4482	4493	22124	18763	13725	19521	24109	14748	3291	139092
Ovalipes ocellatus	1845	4337	3050	11258	17912	22686	31952	17574	4118	2833	117565
Cancer irroratus	830	633	751	681	684	6058	7925	7137	6680	4448	35827
Carcinus maenus	989	781	656	765	912	3732	2533	6687	5647	2717	25419
Callinectes sapidus	939	437	928	1901	1963	1802	1433	1553	1020	732	12708
Homarus americanus	1141	544	508	718	761	1715	1967	1504	1167	501	10526
Libinia spp.	1598	526	157	119	235	795	865	1360	1484	406	7545
Neopanope texana	0	35	35	28	25	167	785	626	1496	242	3439
Limulus polyphemus	327	59	45	64	89	184	142	162	164	0	1236
Squilla empusa	53	67	6	2	193	34	29	178	3	3	568
Pagurus spp.	227	45	21	4	91	45	18	14	12	7	484
Cancer borealis	18	6	7	100	26	37	25	42	2	0	263
Upogebia affinis	0	0	0	0	0	0	14	128	0	0	142
Argopecten irradians	0	0	0	0	0	5	2	50	38	0	95
Illex illecebrosus	0	19	10	0	0	29	7	5	6	0	76
Penaeus aztecus	0	0	0	0	4	2	47	0	0	7	60
Callinassa atlanticus	0	0	0	0	0	0	0	34	0	0	34
Callinectes similis	0	0	0	0	0	7	18	0	0	0	25
Lunatia heros	0	0	2	0	2	0	7	0	0	0	11
Aplysia wilcoxi	0	0	0	0	2	0	0	0	0	0	2
Hexapanopeus angustifrons	1	0	0	0	0	0	0	0	0	0	1
Total	21804	11971	10669	37764	41662	51023	67290	61163	36585	15187	355118

Appendix X. Trawl catch by year (1976-1985).

	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
Number of samples	908	936	936	936	972	935	936	936	936	936	9367
Fish Taxa											
Pseudopleuronectes americanus	7875	5752	6055	10694	12378	13124	13517	16799	14027	8869	109090
Stenotomus chrysops	2209	4040	2556	4052	3882	3401	4878	5286	4109	2732	37145
Scophthalmus aquosus	1618	1259	736	1408	2133	1549	2745	2970	2339	1951	18708
Anchoa spp.	980	580	2223	15	113	577	39	88	178	9997	14790
Raja spp.	875	541	409	404	728	819	1819	2602	2067	2402	12666
Gadidae	29	231	647	438	391	6627	1424	429	455	698	11369
Menidia spp.	2151	1224	1060	2059	1003	356	427	635	348	465	9728
Tautogolabrus adspersus	1009	1032	359	1381	981	825	561	412	246	143	6949
Myoxocephalus aenaeus	191	276	591	316	458	866	788	904	595	498	5483
Prionotus spp.	370	333	105	341	389	535	1176	411	369	317	4346
Paralichthys dentatus	309	149	89	79	120	232	244	266	1929	210	3627
Merluccius bilinearis	385	141	102	169	533	217	391	135	109	175	2357
Urophycis spp.	194	97	55	102	161	215	356	626	216	231	2253
Tautoga onitis	251	292	246	283	138	235	228	159	110	136	2078
Hemitripterus americanus	31	41	35	82	223	347	381	581	262	72	2055
Microgadus tomcod	19	25	40	49	125	279	1147	132	90	85	1991
Gasterosteus aculeatus	19	22	13	103	38	192	116	256	940	199	1898
Pholis gunnellus	24	147	62	100	145	301	206	290	130	156	1561
Syngnathus fuscus	49	34	58	71	127	288	158	241	262	181	1469
Peprilus triacanthus	37	42	408	173	46	69	182	244	17	134	1352
Osmerus mordax	121	242	102	39	110	49	62	83	221	319	1348
Apeltes quadracus	19	7	5	24	32	196	764	77	22	119	1265
Gadus morhua	3	29	3	2	0	1	26	255	278	348	945
Myoxocephalus octodecemspinosus	43	12	23	92	42	110	117	126	34	24	623
Etropus microstomus	52	6	3	3	26	80	54	88	96	124	532
Paralichthys oblongus	31	8	21	7	53	31	109	54	87	58	459
Ammodytes americanus	1	4	60	127	37	117	14	19	10	19	408
Alosa pseudoharengus	9	38	243	9	18	11	7	6	23	4	368
Opsanus tau	102	22	6	17	32	35	21	28	18	34	315
Centropristis striata	34	9	3	4	10	63	22	39	26	66	276
Anguilla rostrata	20	17	7	6	10	35	28	26	22	32	203
Pollachius virens	1	3	0	19	0	5	33	36	87	19	203
Cyclopterus lumpus	2	17	13	28	56	11	1	14	0	29	171
Liparis spp.	8	8	26	12	19	12	35	9	15	11	155
Alosa sapidissima	32	6	2	3	42	9	3	28	1	0	126
Clupeidae	2	1	0	0	0	0	0	0	0	110	113
Clupea harengus	5	1	9	13	0	1	0	1	6	67	103
Cynoscion regalis	9	21	4	2	2	45	4	3	1	5	96
Sphoeroides maculatus	17	10	1	0	9	14	16	15	6	8	96
Mustelis canis	2	5	45	11	1	5	4	6	0	2	81
Alosa aestivalis	5	0	12	10	13	1	0	8	11	5	65
Brevoortia tyrannus	1	4	11	10	2	1	0	0	1	34	64
Limanda ferruginea	0	7	5	6	1	9	10	5	1	3	47
Monacanthus hispidus	3	6	8	4	0	0	8	1	8	9	47
Morone americana	4	7	15	3	11	3	1	1	0	0	45
Macrozoarces americanus	0	5	8	8	3	3	0	2	2	3	34
Gobiidae	3	0	0	0	4	0	0	3	2	13	25
Fistularia tabacaria	2	3	0	0	3	0	1	0	8	1	18

Appendix X. (continued)

Fish Taxa	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
Leiostomus xanthurus	3	2	6	0	0	0	0	2	0	2	15
Gasterosteidae	.	.	.	.	13	.	.	.	.	.	13
Pomatomus saltatrix	1	1	0	2	1	2	3	3	0	0	13
Hippocampus spp.	0	0	0	0	0	0	0	1	4	6	11
Aluterus schoepfi	0	2	2	1	1	0	0	1	1	2	10
Dactylopterus volitans	3	0	0	0	0	1	3	1	0	1	9
Synodus foetens	0	1	4	0	0	3	1	0	0	0	9
Pungitius pungitius	0	0	0	0	0	1	2	0	5	0	8
Fundulus spp.	0	0	0	0	0	4	3	0	0	0	7
Priacanthus cruentatus	0	0	0	0	0	1	0	2	3	1	7
Menticirrhus saxatilis	0	1	0	1	0	3	1	0	0	0	6
Morone saxatilis	0	0	2	1	0	1	1	0	0	1	6
Caranx crysos	0	0	0	0	1	0	1	0	1	2	5
Lophius americanus	2	0	0	0	1	0	0	2	0	0	5
Trinectes maculatus	3	1	0	0	0	0	0	0	0	1	5
Ulvaria subbifurcata	1	0	2	0	0	1	1	0	0	0	5
Conger oceanicus	1	0	0	0	0	1	0	0	1	1	4
Pristigenys alta	0	0	0	0	0	1	0	0	2	1	4
Sphyraena borealis	0	0	0	0	0	0	0	1	1	2	4
Trachurus lathami	0	0	0	3	1	0	0	0	0	0	4
Gasterosteus wheatlandi	.	.	.	.	.	.	1	1	1	.	3
Lactophrys spp.	0	0	0	0	0	0	0	0	3	0	3
Mullus auratus	0	0	1	0	0	0	2	0	0	0	3
Ophidion marginatum	0	0	0	0	0	0	0	0	0	3	3
Priacanthus arenatus	0	0	0	0	0	0	0	0	2	1	3
Selene vomer	1	2	0	0	0	0	0	0	0	0	3
Alosa mediocris	1	0	0	0	0	1	0	0	0	0	2
Caranx hippos	0	0	0	0	0	0	1	0	0	1	2
Chaetodon ocellatus	0	0	0	0	1	0	0	1	0	0	2
Decapterus macarellus	0	0	0	0	0	0	0	0	2	0	2
Mugil cephalus	0	0	1	0	0	0	0	1	0	0	2
Scomber scombrus	0	1	0	1	0	0	0	0	0	0	2
Squalus acanthias	0	0	0	0	0	0	1	0	1	0	2
Acipenser oxyrhynchus	0	0	0	0	1	0	0	0	0	0	1
Aulostomus maculatus	1	0	0	0	0	0	0	0	0	0	1
Bairdiella chrysoura	0	0	0	0	0	0	0	1	0	0	1
Bothus ocellatus	0	0	0	0	0	0	1	0	0	0	1
Dasyatis centroura	0	0	0	0	0	0	0	0	1	0	1
Enchelyopus cimbrius	0	0	0	0	0	0	1	0	0	0	1
Melanogrammus aeglefinus	0	0	0	0	0	0	0	1	0	0	1
Myliobatis freminvillei	0	0	0	0	0	0	1	0	0	0	1
Salmo trutta	0	0	0	0	1	0	0	0	0	0	1
Scyliorhinus retifer	1	0	0	0	0	0	0	0	0	0	1
Selar crumenophthalmus	0	0	0	0	0	0	0	0	0	1	1
Selene setapinnis	0	0	0	0	0	0	0	0	1	0	1
Trachinocephalus myops	0	0	0	0	0	0	0	0	1	0	1
Trachinotus falcatus	0	0	0	0	0	0	0	0	1	0	1
Upeneus parvus	0	0	0	0	0	0	0	0	0	1	1
Total	19174	16767	16502	22787	24669	31921	32147	34417	29815	31144	259343

Appendix X. (continued)

Invertebrate Taxa	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
Libinia spp.	1268	3832	339	331	1991	2177	1822	2856	2932	3206	20754
Carcinus maenus	276	86	58	81	805	1643	3152	1203	2539	2965	12808
Cancer irroratus	91	91	193	142	469	1713	2112	5530	828	996	12165
Loligo pealei	520	1062	421	563	860	1838	668	1943	891	1134	9900
Lunatia heros	0	0	0	0	1626	771	220	453	494	1157	4721
Ovalipes ocellatus	37	116	110	215	680	392	1168	647	291	189	3845
Homarus americanus	103	141	150	118	213	572	1170	623	407	331	3828
Argopecten irradians	147	315	157	47	445	506	172	326	370	636	3121
Libinia spp.	5	0	0	0	0	415	119	102	1329	1009	2979
Pagurus pollicaris	0	0	0	0	207	312	367	1098	274	451	2709
Neopanope texana	0	0	0	0	32	161	281	282	188	317	1261
Asterias forbesi	0	0	0	0	503	125	23	0	4	2	657
Callinectes sapidus	56	17	3	15	138	42	59	82	50	75	537
Limulus polyphemus	0	0	0	0	39	45	58	47	40	39	268
Busycon canaliculatum	0	0	0	0	52	65	43	42	39	26	267
Cancer borealis	48	42	0	0	31	1	0	0	0	1	123
Cancer spp.	76	7	1	1	2	0	0	0	0	0	87
Busycon carica	0	0	0	0	0	11	3	20	24	18	76
Polinices duplicata	0	0	0	0	6	5	3	11	2	7	34
Squilla empusa	0	0	0	0	6	4	1	4	3	4	22
Callinectes similis	0	0	0	0	0	0	11	0	0	5	16
Henricia sanguinolenta	0	0	0	0	4	0	0	0	0	0	4
Illex illecebrosus	0	0	0	0	0	0	0	0	0	1	1
Total	2622	5709	1432	1513	8109	10383	11333	15167	9376	11559	77203

Appendix XI. Trawl catch by station (1976-1985).

	JC	NR	NB	TT	BR	IN	Total
Number of samples	1563	1559	1563	1563	1563	1556	9367
Fish Taxa							
Pseudopleuronectes americanus	10786	33899	15760	18697	12923	17025	109090
Stenotomus chrysops	2592	221	15841	5710	3124	9657	37145
Scophthalmus aquosus	1064	1421	1676	2604	9876	2067	18708
Anchoa spp.	718	240	10263	293	15	3261	14790
Raja spp.	689	10	2088	3259	5130	1490	12666
Gadidae	2196	584	4435	1744	326	2084	11369
Menidia spp.	2186	2380	1312	493	152	3205	9728
Tautogolabrus adspersus	1412	297	495	260	377	4108	6949
Myoxocephalus aenaeus	846	2075	329	426	647	1160	5483
Prionotus spp.	72	424	346	905	2144	455	4346
Paralichthys dentatus	493	588	402	1592	133	419	3627
Merluccius bilinearis	139	3	328	306	1116	465	2357
Urophycis spp.	301	22	212	230	1154	334	2253
Tautoga onitis	442	427	252	177	222	558	2078
Hemitripterus americanus	443	81	403	286	444	398	2055
Microgadus tomcod	638	466	484	71	27	305	1991
Gasterosteus aculeatus	1254	612	8	10	6	8	1898
Pholis gunnellus	851	187	164	98	13	248	1561
Syngnathus fuscus	450	668	103	55	61	132	1469
Peprilus triacanthus	19	3	178	373	730	49	1352
Osmerus mordax	846	227	90	57	36	92	1348
Apeltes quadracus	48	1214	0	1	1	1	1265
Gadus morhua	203	18	244	110	79	291	945
Myoxocephalus octodecemspinosus	3	0	20	80	505	15	623
Etropus microstomus	58	4	51	78	263	78	532
Paralichthys oblongus	0	2	45	6	390	16	459
Ammodytes americanus	19	94	4	28	257	6	408
Alosa pseudoharengus	7	59	16	7	46	233	368
Opsanus tau	5	299	0	0	0	11	315
Centropristis striata	20	35	26	16	22	157	276
Anguilla rostrata	35	146	0	14	2	6	203
Pollachius virens	51	8	7	11	8	118	203
Cyclopterus lumpus	105	4	11	6	2	43	171
Liparis spp.	19	1	32	26	53	24	155
Alosa sapidissima	8	17	50	9	20	22	126
Clupeidae	0	1	0	1	0	111	113
Clupea harengus	63	3	15	4	14	4	103
Cynoscion regalis	18	0	19	8	36	15	96
Sphoeroides maculatus	12	56	9	1	12	6	96
Mustelis canis	4	1	39	3	30	4	81
Alosa aestivalis	1	22	13	6	13	10	65
Brevoortia tyrannus	0	48	12	1	0	3	64
Limanda ferruginea	0	0	0	4	43	0	47
Monacanthus hispidus	15	1	7	5	10	9	47
Morone americana	6	11	4	1	6	17	45
Macrozoarces americanus	0	0	0	1	32	1	34
Gobiidae	2	23	0	0	0	0	25
Fistularia tabacaria	14	1	0	0	0	3	18

Appendix XI. (continued)

Fish Taxa	JC	NR	NB	TT	BR	IN	Total
Leiostomus xanthurus	2	0	7	0	3	3	15
Gasterosteidae	2	11	.	.	.	.	13
Pomatomus saltatrix	2	3	2	0	5	1	13
Hippocampus spp.	4	3	1	1	0	2	11
Aluterus schoepfi	5	0	0	1	1	3	10
Dactylopterus volitans	0	7	0	0	0	2	9
Synodus foetens	0	3	0	2	4	0	9
Pungitius pungitius	5	2	0	0	0	1	8
Fundulus spp.	0	7	0	0	0	0	7
Priacanthus cruentatus	1	0	1	2	0	3	7
Menticirrhus saxatilis	0	2	1	2	0	1	6
Morone saxatilis	0	6	0	0	0	0	6
Caranx crysos	0	0	2	0	1	2	5
Lophius americanus	0	0	0	1	4	0	5
Trinectes maculatus	5	0	0	0	0	0	5
Ulvaria subbifurcata	3	0	1	1	0	0	5
Conger oceanicus	1	1	0	0	2	0	4
Pristigenys alta	1	0	0	1	1	1	4
Sphyraena borealis	4	0	0	0	0	0	4
Trachurus lathami	1	0	3	0	0	0	4
Gasterosteus wheatlandi	3	.	.	.	.	.	3
Lactophrys spp.	2	1	0	0	0	0	3
Mullus auratus	1	0	0	0	0	2	3
Ophidion marginatum	0	0	0	2	1	0	3
Priacanthus arenatus	0	1	0	0	0	2	3
Selene vomer	1	0	1	0	0	1	3
Alosa mediocris	1	0	0	0	1	0	2
Caranx hippos	0	0	0	0	0	2	2
Chaetodon ocellatus	1	1	0	0	0	0	2
Decapterus macarellus	1	0	1	0	0	0	2
Mugil cephalus	1	0	1	0	0	0	2
Scomber scombrus	0	0	1	0	0	1	2
Squalus acanthias	0	0	0	0	2	0	2
Acipenser oxyrhynchus	0	0	1	0	0	0	1
Aulostomus maculatus	1	0	0	0	0	0	1
Bairdiella chrysoura	0	0	0	0	1	0	1
Bothus ocellatus	0	1	0	0	0	0	1
Dasyatis centroura	1	0	0	0	0	0	1
Enchelyopus cimbrius	0	0	0	0	1	0	1
Melanogrammus aeglefinus	1	0	0	0	0	0	1
Myliobatis freminvillei	0	0	1	0	0	0	1
Salmo trutta	0	1	0	0	0	0	1
Scyliorhinus retifer	0	1	0	0	0	0	1
Selar crumenopthalmus	0	0	0	1	0	0	1
Selene setapinnis	0	0	1	0	0	0	1
Trachinocephalus myops	1	0	0	0	0	0	1
Trachinotus falcatus	0	0	0	0	0	1	1
Upeneus parvus	1	0	0	0	0	0	1
Total	29205	46954	55818	38087	40527	48752	259343

Appendix XI. (continued)

Invertebrate Taxa	JC	NR	NB	TT	BR	IN	Total
Libinia spp.	1207	12614	2486	2670	456	1321	20754
Carcinus maenus	464	12265	28	13	11	27	12808
Cancer irroratus	1092	1391	2194	5135	1073	1280	12165
Loligo pealei	1021	140	1823	1783	2892	2241	9900
Lunatia heros	11	1	374	1481	2840	14	4721
Ovalipes ocellatus	416	2565	46	424	382	12	3845
Homarus americanus	1294	341	599	224	288	1082	3828
Argopecten irradians	139	2973	6	1	0	2	3121
Pagurus pollicaris	155	76	351	424	1565	138	2709
Neopanope texana	339	848	21	16	4	33	1261
Asterias forbesi	12	133	19	53	401	39	657
Callinectes sapidus	97	248	110	12	13	57	537
Limulus polyphemus	16	210	4	9	8	21	268
Busycon canaliculatum	24	45	20	79	49	50	267
Cancer borealis	30	14	43	14	11	11	123
Cancer spp.	5	2	47	22	4	7	87
Busycon carica	0	2	13	26	22	13	76
Polinices duplicata	2	3	5	2	10	12	34
Squilla empusa	2	0	4	0	5	11	22
Callinectes similis	0	11	0	0	0	5	16
Henricia sanguinolenta	0	0	0	2	0	2	4
Illex illecebrosus	0	0	0	1	0	0	1
Total	6326	33882	8199	12390	10034	6378	77203

Appendix XII. Seine catch by year (1976-1985).

	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	Total
Number of samples	66	72	72	72	72	72	80	99	174	156	935
Taxa											
Menidia spp.	40620	18179	1178	1233	7764	3418	5408	9007	2330	1342	90479
Fundulus spp.	1746	1174	838	660	943	612	914	870	1659	901	10317
Apeltes quadracus	464	602	254	269	35	109	79	1767	230	107	3916
Cyprinodon variegatus	57	672	41	30	10	352	144	47	33	29	1415
Ammodytes americanus	6	520	16	51	10	318	82	30	21	0	1054
Pungitius pungitius	6	1	11	19	2	3	9	322	10	10	393
Gasterosteus aculeatus	8	151	8	30	2	3	5	49	9	4	269
Syngnathus fuscus	9	3	9	108	6	8	21	9	38	27	238
Pomatomus saltatrix	1	0	1	6	0	2	49	90	19	35	203
Alosa pseudoharengus	0	0	0	0	0	0	0	1	93	0	94
Gadidae	2	0	9	1	1	24	21	2	18	10	88
Pseudopleuronectes americanus	4	6	4	1	2	9	2	1	18	38	85
Mugil cephalus	0	4	3	18	46	1	4	0	5	0	81
Gasterosteus wheatlandi	.	.	.	.	.	8	3	5	5	18	39
Brevoortia tyrannus	0	0	17	0	4	0	7	1	0	8	37
Anguilla rostrata	11	2	14	4	1	1	1	0	1	0	35
Myoxocephalus aenaeus	3	2	1	2	0	0	3	1	3	3	18
Anchoa spp.	0	0	0	0	2	0	7	2	1	0	12
Mugil curema	0	0	0	0	0	0	0	1	9	0	10
Alosa aestivalis	2	6	0	0	0	0	0	0	0	0	8
Lucania parva	1	2	0	0	0	0	0	2	0	1	6
Tautogolabrus adspersus	0	0	2	0	0	0	3	0	1	0	6
Sphoeroides maculatus	0	0	0	1	0	0	1	0	0	3	5
Tautoga onitis	0	0	0	0	0	0	4	0	0	0	4
Trachinotus falcatus	0	0	1	0	3	0	0	0	0	0	4
Caranx hippos	0	0	1	0	0	1	0	0	0	1	3
Alosa sapidissima	1	0	0	0	0	0	0	0	0	1	2
Clupea harengus	0	0	0	0	0	0	2	0	0	0	2
Menticirrhus saxatilis	1	0	1	0	0	0	0	0	0	0	2
Osmerus mordax	0	0	0	0	0	0	0	0	0	2	2
Peprilus triacanthus	0	0	0	0	0	0	1	0	1	0	2
Strongylura marina	0	0	0	0	0	1	1	0	0	0	2
Clupeidae	1	0	0	0	0	0	0	0	0	0	1
Cynoscion regalis	0	0	0	0	0	0	0	0	0	1	1
Pholis gunnellus	0	0	0	0	0	0	0	0	0	1	1
Prionotus spp.	0	0	0	0	0	0	0	0	0	1	1
Scophthalmus aquosus	0	0	0	0	0	0	0	0	0	1	1
Urophycis spp.	0	0	0	0	0	0	0	0	1	0	1
Total	42943	21324	2409	2433	8831	4870	6771	12207	4505	2544	108837

Appendix XIII. Seine catch by station (1976-1985).

	JC	WP	GN	Total
Number of samples	300	314	321	935
Taxa				
Menidia spp.	72783	9879	7817	90479
Fundulus spp.	7747	1502	1068	10317
Apeltes quadracus	3896	6	14	3916
Cyprinodon variegatus	625	767	23	1415
Ammodytes americanus	2	197	855	1054
Pungitius pungitius	322	67	4	393
Gasterosteus aculeatus	227	19	23	269
Syngnathus fuscus	45	34	159	238
Pomatomus saltatrix	141	7	55	203
Alosa pseudoharengus	5	89	0	94
Gadidae	58	24	6	88
Pseudopleuronectes americanus	19	7	59	85
Mugil cephalus	55	2	24	81
Gasterosteus wheatlandi	12	13	14	39
Brevoortia tyrannus	2	6	29	37
Anguilla rostrata	32	2	1	35
Myoxocephalus aenaeus	4	7	7	18
Anchoa spp.	11	1	0	12
Mugil curema	10	0	0	10
Alosa aestivalis	1	5	2	8
Lucania parva	4	0	2	6
Tautogolabrus adspersus	5	1	0	6
Sphoeroides maculatus	0	0	5	5
Tautoga onitis	4	0	0	4
Trachinotus falcatus	2	2	0	4
Caranx hippos	2	0	1	3
Alosa sapidissima	0	0	2	2
Clupea harengus	2	0	0	2
Menticirrhus saxatilis	1	0	1	2
Osmerus mordax	0	0	2	2
Peprilus triacanthus	0	1	1	2
Strongylura marina	2	0	0	2
Clupeidae	0	1	0	1
Cynoscion regalis	1	0	0	1
Pholis gunnellus	0	0	1	1
Prionotus spp.	0	1	0	1
Scophthalmus aquosus	0	0	1	1
Urophycis spp.	0	1	0	1
Total	86020	12641	10176	108837

Appendix XIV. Time-series models used to describe the log-transformed density (no./500 m^3) of larval *Ammodytes americanus* collected at stations EN and NB and analytical summary.

Station = EN

Model[a] : $Z_t = B_1S(1 + B_2SIN(tK_6) - B_3COS(tK_6) - B_4SIN(tK_4) + B_5COS(tK_4)) + A_1 + A_2$

Model summary statistics: SSE = 627.5, df = 515, MSE = 1.22, R^2 = 0.74

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast: Above	Below
1976	44.2	45	0.98	.	.
1977	110.4	44	2.51*	37.3	13.4
1978	160.4	45	3.57*	43.0	16.4
1979	57.0	44	1.30	.	.
1980	53.8	44	1.22	.	.
1981	69.8	44	1.59	.	.
1982	22.9	44	0.52	.	.
1983	27.7	44	0.63	.	.
1984	30.2	45	0.67	.	.
1985	51.1	44	1.16	.	.

Station = NB

Model[a] : $Z_t = B_1S(1 + B_2SIN(tK_6) - B_3COS(tK_6) - B_4SIN(tK_4) + B_5COS(tK_4)) + A_1 + A_2 + A_4 + M_{10}$

Model summary statistics: SSE = 170.0, df = 355, MSE = 0.48, R^2 = 0.92

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast: Above	Below
1979	17.0	42	0.41	.	.
1980	18.3	42	0.44	.	.
1981	31.6	42	0.75*	19.1	0.7
1982	29.3	42	0.70*	0.2	19.0
1983	8.3	42	0.20	.	.
1984	16.4	43	0.38	.	.
1985	49.2	42	1.17*	1.0	23.6

a Z_t = mean of LOG((no./500 m^3) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
S = dummy variable for season (see Table 7)

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p > 0.05$)

Appendix XV. Time-series models used to describe the log-transformed catch (no./24 h) of *Anchoa* spp. impinged at MNPS Unit 2 and analytical summary.

Model[a] : $Z_t = B_1 FU2(1 - B_2 SIN(tK_{12}) - B_3 COS(tK_{12}) - B_4 SIN(tK_6) + B_5 COS(tK_6)) - A_1 - A_2 - A_3 + A_6 + A_7 - A_9$

Model summary statistics: SSE = 1729.8, df = 510, MSE = 3.39, R^2 = 0.69

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast: Above	Below
1976	90.3	41	2.20	.	.
1977	86.5	40	2.16	.	.
1978	146.9	40	3.67	.	.
1979	82.3	40	2.06	.	.
1980	195.0	41	4.76*	37.3	36.2
1981	164.9	40	4.12	.	.
1982	122.4	40	3.06	.	.
1983	331.5	40	8.29*	61.6	34.3
1984	250.1	40	6.25*	64.6	30.3
1985	259.4	40	6.49*	23.5	67.9

a Z_t = mean of LOG((no./24 h) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
FU2 = water flow through Unit 2

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p > 0.05$)

Appendix XVI. Time-series models used to describe the log-transformed density (no./500 m^3) of *Anchoa* spp. eggs collected at EN and analytical summary.

Model[a] : $Z_t = B_1S(B_2SIN(tK_{12}) - B_3COS(tK_{12}) + B_4SIN(tK_6) + B_5COS(tK_6)) + A_1 + A_4 + M_7$

Model summary statistics: SSE = 279.3, df = 357, MSE = 0.83, R^2 = 0.84

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast: Above	Below
1979	25.9	44	0.59	.	.
1980	35.3	44	0.80	.	.
1981	17.3	44	0.39	.	.
1982	26.9	44	0.61	.	.
1983	64.1	44	1.46*	19.6	8.1
1984	77.1	45	1.71*	19.6	9.5
1985	50.7	44	1.15*	2.5	20.0

a Z_t = mean of LOG((no./500 m^3) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
S = dummy variable for season (see Table 7)

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p > 0.05$)

Appendix XVII. Time-series models used to describe the log-transformed density (no./500 m^3) of larval *Anchoa* spp. collected at EN and NB and analytical summary.

Station = EN

Model[a] : $Z_t = B_1S(1 + B_2SIN(tK_6) + B_3COS(tK_6) + B_4SIN(tK_1) \quad B_5COS(tK_3)) + A_1 + M_3 + M_9$

Model summary statistics: SSE = 336.5, df = 514, MSE = 0.65, R^2 = 0.93

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast: Above	Below
1976	18.0	44	0.41	.	.
1977	18.6	43	0.43	.	.
1978	79.7	44	1.81*	8.7	24.1
1979	22.8	43	0.53	.	.
1980	24.9	43	0.58	.	.
1981	33.6	43	0.78	.	.
1982	27.0	43	0.63	.	.
1983	35.5	43	0.83	.	.
1984	46.9	44	1.07*	4.0	24.7
1985	29.4	43	0.68	.	.

Station = NB

Model[a] : $Z_t = B_1S(1 \quad B_2SIN(tK_4) + B_3COS(tK_4) \quad B_4SIN(tK_3) + B_5COS(tK_3))$

Model summary statistics: SSE = 269.2, df = 359, MSE = 0.75, R^2 = 0.92

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast: Above	Below
1979	32.2	46	0.70	.	.
1980	34.9	46	0.76	.	.
1981	55.9	46	1.21*	13.1	15.6
1982	16.2	46	0.35	.	.
1983	29.2	46	0.64	.	.
1984	59.6	47	1.27*	1.6	25.4
1985	41.2	46	0.89	.	.

a Z_t = mean of LOG((no./500 m^3) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
S = dummy variable for season (see Table 7)

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p = 0.05$)

Appendix XVIII. Time-series models used to describe the log-transformed catch (no./24 h) of *Gasterosteus* spp. impinged at MNPS Unit 2 and analytical summary.

Model[a] : $Z_t = B_1 FU2(1 + B_2 SIN(tK_{12}) + B_3 COS(tK_{12}) - B_4 SIN(tK_4) + B_5 COS(tK_4)) - A_1 - A_2 + A_5$

Model summary statistics: SSE = 1546.5, df = 513, MSE = 3.01, R^2 = 0.82

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast: Above	Below
1976	69.8	44	1.59	.	.
1977	64.3	43	1.50	.	.
1978	141.7	43	3.30	.	.
1979	137.5	43	3.20	.	.
1980	120.0	44	2.73	.	.
1981	106.1	43	2.47	.	.
1982	176.3	43	4.10	.	.
1983	106.1	43	2.47	.	.
1984	360.8	43	8.39*	41.9	58.7
1985	263.4	43	6.12*	24.3	58.2

a Z_t = mean of LOG((no./24 h) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
$FU2$ = water flow through Unit 2

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p > 0.05$)

Appendix XIX. Time-series models used to describe the log-transformed seine catch (no./100 m) of *Menidia* spp. collected at stations GN, JC and WP and analytical summary.

Station – GN

Model[a] : $Z_t = B_1S(1 - B_2SIN(tK_{12}) + B_3COS(tK_{12}) + B_4SIN(tK_6) + B_5COS(tK_6))$

Model summary statistics: SSE – 265.3, df = 194, MSE – 1.37, R^2 – 0.88

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast	
				Above	Below
1969	13.7	2	6.83*	7.4	4.1
1970	7.6	6	1.26	.	.
1971	31.3	6	5.22*	8.4	4.8
1972	28.2	6	4.70*	6.5	5.7
1973	9.6	6	1.60	.	.
1974	26.3	6	4.38*	12.2	1.6
1975	16.3	6	2.71	.	.
1976	24.1	6	4.02*	2.9	9.7
1977	6.1	6	1.02	.	.
1978	10.3	6	1.71	.	.
1979	4.8	6	0.80	.	.
1980	9.0	6	1.49	.	.
1981	20.4	6	3.39*	4.3	4.6
1982	7.0	6	1.17	.	.
1983	24.1	6	4.01*	3.3	9 .0
1984	21.9	6	3.65*	3.8	8.3
1985	4.8	6	0.80	.	.

Station – JC

Model[a] : $Z_t = B_1S(1 + B_2SIN(tK_6) + B_3COS(tK_6))$

Model summary statistics: SSE = 150.4, df = 196, MSE = 0.77, R^2 = 0.83

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast	
				Above	Below
1969	5.9	4	1.47	.	.
1970	10.5	8	1.31	.	.
1971	12.6	8	1.58*	3.9	2.4
1972	4.1	8	0.51	.	.
1973	16.9	8	2.11*	5.0	3.3
1974	8.7	8	1.08	.	.
1975	13.3	8	1.66*	0.7	6.5
1976	3.8	8	0.47	.	.
1977	13.8	8	1.72*	4.3	4.6
1978	4.6	8	0.58	.	.
1979	3.9	8	0.49	.	.
1980	8.9	8	1.12	.	.
1981	9.3	8	1.19	.	.
1982	6.9	8	0.86	.	.
1983	9.6	8	1.19	.	.
1984	10.0	8	1.25	.	.
1985	7.4	8	0.92	.	.

Appendix XIX. (continued)

Station = WP

Model[a] : $Z_t = B_1S(1 + B_2SIN(tK_{12}) + B_3COS(tK_{12}) + B_4SIN(tK_6) - B_5COS(tK_6))$

Model summary statistics: SSE = 15.6, df = 194, MSE = 0.08, R^2 = 0.81

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast	
				Above	Below
1969	0.5	2	0.23*	0.1	1.4
1970	0.7	6	0.11	.	.
1971	0.6	6	0.09	.	.
1972	1.0	6	0.17	.	.
1973	0.3	6	0.05	.	.
1974	1.3	6	0.22*	1.7	0.2
1975	1.6	6	0.26*	1.2	1.1
1976	1.1	6	0.19*	0.2	2.2
1977	0.5	6	0.08	.	.
1978	1.0	6	0.17	.	.
1979	0.7	6	0.11	.	.
1980	0.5	6	0.08	.	.
1981	0.9	6	0.15	.	.
1982	1.4	6	0.23*	1.7	1.0
1983	1.4	6	0.21*	1.8	0.9
1984	1.1	6	0.18*	1.5	1.5
1985	1.2	6	0.20*	2.3	0.2

a Z_t = mean of LOG((no./100 m) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
S = dummy variable for season (see Table 7)

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p > 0.05$)

Appendix XX. Time-series models used to describe the log-transformed catch (no./24 h) of *Menidia* spp. impinged at MNPS Unit 2 and analytical summary.

Model[a] : $Z_t = B_1 FU2(1 + B_2 SIN(tK_{12}) + B_3 COS(tK_{12})) - A_1 \quad A_2 + A_{11} - A_{12}$

Model summary statistics: SSE = 1349.4, df = 514, MSE = 2.63, R^2 = 0.75

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast	
				Above	Below
1976	82.9	45	1.84	.	.
1977	76.7	44	1.74	.	.
1978	129.8	44	2.95	.	.
1979	102.3	44	2.33	.	.
1980	179.5	45	3.99*	42.1	23.6
1981	153.2	44	3.48	.	.
1982	133.7	44	3.04	.	.
1983	88.8	44	2.02	.	.
1984	238.0	44	5.41*	15.1	69.2
1985	164.5	44	3.74*	20.9	44.7

a Z_t = mean of LOG((no./24 h) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
$FU2$ = water flow through Unit 2

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p > 0.05$)

Appendix XXI. Time-series models used to describe the log-transformed catch (no./24 h) of *Microgadus tomcod* impinged at MNPS Unit 2 and analytical summary.

Model[a] : $Z_t = B_1 FU2\left(1 + B_2 SIN(tK_{12}) + B_3 COS(tK_{12}) - B_4 SIN(tK_6) + B_5 COS(tK_6)\right) - A_1 - A_2$

Model summary statistics: SSE = 1538.0, df = 514, MSE = 2.99, R^2 = 0.69

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast: Above	Below
1976	34.6	45	0.77	.	.
1977	64.9	44	1.48	.	.
1978	107.8	44	2.45	.	.
1979	208.8	44	4.75*	46.6	36.7
1980	91.0	45	2.02	.	.
1981	196.0	44	4.45*	48.9	28.7
1982	196.6	44	4.47*	49.7	34.9
1983	103.5	44	2.35	.	.
1984	351.4	44	7.99*	62.2	56.1
1985	182.9	44	4.16*	23.3	52.3

a Z_t = mean of LOG((no./24 h) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
FU2 = water flow through Unit 2

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p > 0.05$)

Appendix XXII. Time-series models used to describe the log-transformed catch (no./24 h) of *Myoxocephalus aenaeus* impinged at MNPS Unit 2 and analytical summary.

Model[a] : $Z_t = B_1 FU2(1 + B_2 SIN(tK_{12}) + B_3 COS(tK_{12})) - A_1 - A_2$

Model summary statistics: SSE = 1804.6, df = 516, MSE = 3.50, R^2 = 0.83

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast	
				Above	Below
1976	124.7	47	2.65	.	.
1977	110.2	46	2.40	.	.
1978	151.8	46	3.30	.	.
1979	170.5	46	3.68	.	.
1980	139.1	47	2.96	.	.
1981	126.3	46	2.75	.	.
1982	142.0	46	3.09	.	.
1983	189.2	46	4.11	.	.
1984	305.7	46	6.65*	57.6	45.3
1985	345.7	46	7.51*	59.7	46.9

a Z_t = mean of LOG((no./24 h) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
$FU2$ = water flow through Unit 2

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p > 0.05$)

Appendix XXIII. Time-series models used to describe the log-transformed density (no./500 m^3) of larval *Myoxocephalus aenaeus* collected at stations EN and NB and analytical summary.

Station = EN

Model[a] : $Z_t = B_1 S(B_2 SIN(tK_{12}) + B_3 COS(tK_{12}) - B_4 SIN(tK_4) - B_5 COS(tK_4)) + A_1 - A_5 - M_2 - M_6$

Model summary statistics: SSE = 111.4, df = 514, MSE = 0.22, R^2 = 0.93

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast	
				Above	Below
1976	13.1	44	0.30*	0.5	15.2
1977	15.4	43	0.36*	4.6	10.7
1978	7.5	44	0.17	.	.
1979	3.6	43	0.08	.	.
1980	5.7	43	0.13	.	.
1981	19.2	43	0.45*	9.4	5.2
1982	10.0	43	0.23	.	.
1983	19.9	43	0.46*	12.3	4.4
1984	9.6	44	0.22	.	.
1985	7.3	43	0.17	.	.

Station = NB

Model[a] : $Z_t = B_1 S(B_2 SIN(tK_{12}) + B_3 COS(tK_{12}) - B_4 SIN(tK_4) + B_5 COS(tK_4))$

Model summary statistics: SSE = 62.1, df = 360, MSE = 0.17, R^2 = 0.93

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast	
				Above	Below
1979	8.1	47	0.17	.	.
1980	5.8	47	0.12	.	.
1981	10.5	47	0.22	.	.
1982	3.6	47	0.07	.	.
1983	12.1	47	0.26*	6.5	5.8
1984	10.7	48	0.22	.	.
1985	11.4	47	0.24*	9.0	4.5

a Z_t = mean of LOG((no./500 m^3) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
S = dummy variable for season (see Table 7)

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p > 0.05$)

Appendix XXIV. Time-series models used to describe the log-transformed density (no./500 m^3) of *Tautoga onitis* eggs collected at station EN and analytical summary.

Model[a] : $Z_t = B_1S(-B_2SIN(tK_{12}) - B_3COS(tK_{12}) + B_4SIN(tK_6) + B_5COS(tK_6)) + A_1 + M_5$

Model summary statistics: SSE = 163.8, df = 358, MSE = 0.46, R^2 = 0.97

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast	
				Above	Below
1979	14.2	45	0.32	.	.
1980	20.9	45	0.46	.	.
1981	38.6	45	0.86*	9.5	12.5
1982	19.0	45	0.42	.	.
1983	10.9	45	0.24	.	.
1984	35.5	46	0.77*	12.3	12.1
1985	24.7	45	0.55	.	.

a Z_t = mean of LOG((no./500 m^3) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
S = dummy variable for season (see Table 7)

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p > 0.05$)

Appendix XXV. Time-series models used to describe the log-transformed density (no./500 m^3) of larval *Tautoga onitis* collected at stations EN and NB and analytical summary.

Station = EN

Model[a] : $Z_t = B_1 S(-B_2 SIN(tK_{12}) - B_3 COS(tK_{12}) - B_4 SIN(tK_4) - B_5 COS(tK_4) + B_6 SIN(tK_2) + B_7 COS(tK_2)) + A_1$

Model summary statistics: SSE = 151.3, df = 515, MSE = 0.29, R^2 = 0.71

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast: Above	Below
1976	14.0	45	0.32	.	.
1977	8.2	44	0.19	.	.
1978	27.1	45	0.60*	0.5	15.2
1979	6.7	44	0.15	.	.
1980	14.6	44	0.33	.	.
1981	28.1	44	0.64*	13.4	4.0
1982	12.1	44	0.28	.	.
1983	14.5	44	0.33	.	.
1984	19.0	45	0.42*	1.4	13.5
1985	7.0	44	0.16	.	.

Station = NB

Model[a] : $Z_t = B_1 S(-B_2 SIN(tK_{12}) - B_3 COS(tK_{12}) - B_4 SIN(tK_4) - B_5 COS(tK_4) + B_6 SIN(tK_2) + B_7 COS(tK_2))$

Model summary statistics: SSE = 138.5, df = 358, MSE = 0.39, R^2 = 0.85

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast: Above	Below
1979	17.6	45	0.39	.	.
1980	13.9	45	0.31	.	.
1981	25.8	45	0.57*	6.3	11.6
1982	13.7	45	0.30	.	.
1983	23.0	45	0.51	.	.
1984	24.8	46	0.54*	1.8	13.3
1985	19.7	45	0.44	.	.

a Z_t = mean of LOG((no./500 m^3) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
S = dummy variable for season (see Table 7)

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p = 0.05$)

Appendix XXVI. Time-series models used to describe the log-transformed density (no./500 m^3) of *Tautogolabrus adspersus* eggs collected at station EN and analytical summary.

Model[a] : $Z_t = B_1S(B_2SIN(tK_{12}) - B_3COS(tK_{12}) + B_4SIN(tK_4) - B_5COS(tK_4)) + A_1 - A_3$

Model summary statistics: SSE = 227.6, df = 358, MSE = 0.64, R^2 = 0.96

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast	
				Above	Below
1979	15.0	45	0.33	.	.
1980	15.6	45	0.35	.	.
1981	44.9	45	1.00*	7.0	17.5
1982	31.5	45	0.70	.	.
1983	39.1	45	0.87	.	.
1984	44.9	46	0.98*	15.1	11.4
1985	36.6	45	0.81	.	.

a Z_t = mean of LOG((no./500 m^3) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
S = dummy variable for season (see Table 7)

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p > 0.05$)

Appendix XXVII. Time-series models used to describe the log-transformed density (no./500 m^3) of larval *Tautogolabrus adspersus* collected at stations EN and NB and analytical summary.

Station = EN

Model[a] :

$$Z_t = B_1 S\left(- B_2 SIN(tK_{12}) - B_3 COS(tK_{12}) - B_4 SIN(tK_4) - B_5 COS(tK_4) + B_6 SIN(tK_2) + B_7 COS(tK_2)\right) + A_1 + A_2 - A_3 + A_4 - A_5$$

Model summary statistics: SSE = 152.5, df = 511, MSE = 0.30, R^2 = 0.72

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast: Above	Below
1976	3.6	41	0.09	.	.
1977	15.1	40	0.38	.	.
1978	25.8	41	0.63*	1.3	15.0
1979	5.5	40	0.14	.	.
1980	22.7	40	0.57*	14.5	1.1
1981	26.7	40	0.67*	11.9	4.7
1982	8.9	40	0.22	.	.
1983	15.1	40	0.38	.	.
1984	17.2	41	0.42*	1.4	13.3
1985	12.0	40	0.30	.	.

Station = NB

Model[a] : $Z_t = B_1 S\left(- B_2 SIN(tK_{12}) - B_3 COS(tK_{12}) - B_4 SIN(tK_4) - B_5 COS(tK_4)\right)$

Model summary statistics: SSE = 142.3, df = 360, MSE = 0.40, R^2 = 0.85

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast: Above	Below
1979	11.7	47	0.25	.	.
1980	16.9	47	0.36	.	.
1981	28.3	47	0.60*	6.6	12.7
1982	19.6	47	0.42	.	.
1983	24.0	47	0.51	.	.
1984	26.1	48	0.54*	4.1	13.5
1985	15.8	47	0.34	.	.

a Z_t = mean of LOG((no./500 m^3) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
S = dummy variable for season (see Table 7)

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p = 0.05$)

Appendix XXVIII. Time-series models used to describe the log-transformed catch (no./24 h) of *Tautogolabrus adspersus* impinged at MNPS Unit 2 and analytical summary.

Model[a] : $Z_t = B_1 FU2\left(1 - B_2 SIN(tK_{12}) - B_3 COS(tK_{12})\right) - A_1 \cdot A_2 - A_3$

Model summary statistics: SSE = 1819.4, df = 515, MSE = 3.53, R^2 = 0.67

Year	SSE[b]	df	MSE[c]	Sum of deviations from model forecast	
				Above	Below
1976	100.7	46	2.19	.	.
1977	129.5	45	2.88	.	.
1978	174.2	45	3.87	.	.
1979	114.1	45	2.54	.	.
1980	133.8	46	2.91	.	.
1981	133.0	45	2.96	.	.
1982	153.2	45	3.41	.	.
1983	168.9	45	3.75	.	.
1984	446.6	45	9.92*	56.9	80.0
1985	264.9	45	5.89*	36.3	66.9

a Z_t = mean of LOG((no./24 h) + 1)
B_n = regression coefficients
t = time in days
K_m = constant for period of duration m months
A_p = autoregressive coefficients at lag p
M_p = moving average coefficients at lag p
$FU2$ = water flow through Unit 2

b annual component of SSE

c MSE = SSE/df

* Significantly higher than the model MSE, F statistic ($p > 0.05$)

Contents

WINTER FLOUNDER STUDIES 1
INTRODUCTION 1
MATERIALS AND METHODS 3
Adult abundance studies 3
Life history studies 8
Movements and exploitation 13
Stock identification 13
Larval studies 14
Post-larval studies 23
Growth and mortality 25
Impingement 26
Entrainment 26
Impact assessment 27
RESULTS AND DISCUSSION 28
Adult abundance studies 28
Abundance in the Niantic River 28
Harmonic regression models 35
Regional trends in abundance 36
Life history studies 37
Reproduction 37
Age and growth 42
Mortality and survival 49
Food habits 51
Movements and exploitation 53

Stock identification 58
Larval studies 60
Net extrusion studies 60
Diel behavior 61
24-h studies 64
Abundance and distribution 68
Age and growth 79
Tidal export and import 87
Post-larval stage 91
Abundance (age 0) 91
Growth and mortality (age 0) 91
Abundance (age 1) 99
Impingement 102
Abundance 102
Length and sex distribution 105
Fish return sluiceways 106
Entrainment 109
Abundance 109
Survival 113
Impact assessment 115
CONCLUSIONS 117
SUMMARY 118
REFERENCES CITED 123
APPENDICES 139

WINTER FLOUNDER STUDIES

INTRODUCTION

The purpose of this report is to summarize research completed by Northeast Utilities Environmental Laboratory (NUEL) and various consultants on the winter flounder (*Pseudopleuronectes americanus*) prior to 3-unit operation of Millstone Nuclear Power Station (MNPS). Due to its local abundance, this species has been studied intensively since 1973 and considerable data have been collected on its life history, population dynamics, and impact assessment. The large effort devoted to winter flounder studies is related to its importance for the Connecticut sport and commercial fisheries. It is the most valuable commercial finfish in Connecticut and on average makes up about 20% of the total finfish landings. The winter flounder is also one of the most popular marine sport fishes in the state with an estimated annual catch in 1979 of almost 1.4 million fish weighing 412,234 kg (Sampson 1981; Blake and Smith 1984). Its particular life history also makes it potentially susceptible to various types of localized impacts.

The winter flounder ranges from Labrador to Georgia (Leim and Scott 1966) and is one of the most common demersal fishes in inshore waters along the northeastern coast from Nova Scotia to New Jersey (Perlmutter 1947). The population of winter flounder is composed of reproductively isolated stocks which spawn in specific estuaries and coastal areas (Lobell 1939; Perlmutter 1947; Saila 1961). Most adult winter flounder enter natal estuaries in fall and early winter and spawning occurs in late winter and early spring. Females usually mature at age 3 and 4 and males at age 2. Average fecundity is about 500,000 eggs per female with as many as 2.3 million eggs for a 45-cm fish. Winter flounder eggs are demersal and hatch in about 15 d, depending upon water temperature. Small larvae are planktonic and remain in natal estuaries to a great extent, although some may be carried out into open waters by tidal currents. Some of these larvae may return to the estuary on subsequent incoming tides, but the rest are lost from the system. The larval stage lasts about 2 mo, depending upon water temperature. Larger larvae maintain some control over their position by vertical movements and also may spend considerable time on the bottom. Following metamorphosis, most demersal juveniles remain in the estuary in shallow waters. Juveniles are resistant to warm summer water temperatures found there to at least 30.4 °C (Huntsman and Sparks 1924). Immature yearling winter flounder become photonegative and though many remain within the estuary, are usually found in deeper water than young-of-the-year (Pearcy 1962; McCracken 1963). Many adults stay

in estuaries following spawning, while others disperse into deeper waters. By summer, most have left shallow waters as their preferred temperature range is 12-15 °C (McCracken 1963). However, some remain inshore and may escape temperatures above 22.5 °C by burying themselves in cooler bottom sediments (Olla et al. 1969). Adults are omnivores and as opportunistic feeders eat a wide variety of algae and benthic invertebrates. They are sight feeders and are usually active only during the day. Additional details regarding their life history, physiology, behavior, and population dynamics may be found in Klein-MacPhee (1978).

Because winter flounder stocks are localized, NUEL studies have concentrated on the dynamics of the population spawning in the Niantic River to determine if MNPS impacts of impingement and entrainment have caused or would cause changes in local abundance beyond those expected from natural variation. Preliminary field studies to estimate abundance of this population in 1973-74 were expanded in scope in 1975, when surveys using mark and recapture techniques were initiated. An adult abundance survey has been completed each year through the present. In many years, studies of age structure, reproductive activity, growth, survival, movements, early life history, and stock identification have been conducted. For plant impact, impingement and entrainment estimates are available for each year. Data from many of these studies have been used in a predictive mathematical population dynamics model developed by the University of Rhode Island (Saila 1976). This model has formed the basis for all MNPS impact assessments to date, including that for Unit 3 (NUSCo 1983c). However, a stochastic population dynamics model is under development which should more realistically predict population-level effects over the expected duration of MNPS operations. Increased knowledge of larval population dynamics and the stock-recruitment relationship is necessary for the successful application of this model and has been the focus of recent efforts at NUEL.

This report includes a summarization of the data, results, and conclusions for various winter flounder studies from 1973 through the winter 1986 adult abundance survey. Unit 3 began startup tests in fall 1985 and commenced commercial operations in late April of 1986. Therefore, no data from sampling programs (other than the adult survey) after fall 1985 will be presented in this report. These data are from the operational period for Unit 3 and will be included in future reports. Additional information on sampling methodologies, program evaluations, and detailed results and analyses may be found in previous annual reports and documents, including NUSCo (1975, 1976, 1977, 1978a, 1978b, 1979, 1980, 1981a, 1981b, 1982, 1983a, 1983b, 1983c, 1984, 1985, 1986a, 1986b).

MATERIALS AND METHODS

Adult abundance studies

Abundance estimation of the Niantic River population of adult winter flounder has been based on mark and recapture methodologies and details concerning annual surveys from 1973 through 1986 are summarized on Table 1. Fish tagging began in 1973 and 1974, but Niantic River spawners were not specifically targeted; the numbers marked were inadequate for abundance estimation (see NUSCo 1975 for additional details). The 1975 survey design was based on the requirements of the deterministic triple-catch model (Ricker 1958). Fish captured by trawl in various portions of the river were marked by fin clips over a 7-wk period beginning on March 31; recaptures were not remarked (NUSCo 1976). From 1976 through 1981, surveys commenced in early to mid-March and ended during early to mid-May, after all spawning was completed (NUSCo 1977, 1978a, 1979, 1980, 1981a, 1982). Since 1982, each survey started after ice-out in the river from mid to late February and ended in early April, when the proportion of reproductively active females decreased to less than 10% of all females examined for two consecutive weeks (NUSCo 1983a, 1984, 1985, 1986a). In all years since 1975, sampling took place on 2 to 3 d of each week.

In 1976, the Niantic River was subdivided into a number of areas (stations) for each survey (Fig. 1). Stations 1, 2, and 4 were in the navigational channel of the lower to mid-river and 3, 6, 7, and 8 were in the adjacent shallows. After 1979, no tows were made outside of the navigational channel in the lower portion of the river due to an agreement with the East Lyme-Waterford Shellfish Commission to protect bay scallop *(Argopectin irradians)* habitat. In 1983, station 5 in the upper river was subdivided into stations 51-53; station 54 in the upper arm was not established until 1986 (Fig. 2). Some tows in station 51 during 1984-86 along the eastern shore of the upper river extended into the deeper northern portion of former station 6. Tows each week were usually allocated to stations according to station area and the expected abundance of winter flounder; more tows were made where fish were most numerous. In most years, heavy accumulations of macroalgae and detritus that occurred in the deeper portion of station 51 hindered sampling there.

Table 1. Summary of Niantic River adult winter flounder abundance studies from 1973 through 1986.

Year	Dates sampled	Marking method	Size marked	Method of abundance estimation	Comments
1973	Throughout year	Anchor tag	≥ 25*cm*	None	Only 1000 marked in Niantic River -- inadequate for abundance estimate
1974	Throughout year	Anchor tag	≥ 20*cm*	Jolly (1965)	Only 2300 marked in Niantic River. Recaptures not marked.
1975	Mar 31-May 13	Fin clip	≥ 15*cm*	Triple-catch (Ricker 1958), Jolly (1965)	Designed for triple-catch model. Recaptures not marked -- inadequate for Jolly model
1976	Mar 1-May 4	Fin clip, spaghetti tag	≥ 15*cm*	Jolly (1965)	Designed for Jolly model (1976-86). Recaptures in 1976 marked with tags.
1977	Mar 7-May 10	Freeze-brand	≥ 15*cm*	Jolly (1965)	Freeze-brand used to improve marking methodology. Marks indicating both station and week of mark and recapture applied during 1977-79.
1978	Mar 6-May 16	Freeze-brand	≥ 15*cm*	Jolly (1965)	
1979	Mar 12-May 15	Freeze-brand	≥ 15*cm*	Jolly (1965)	Jolly model evaluated.
1980	Mar 17-May 6	Freeze-brand	≥ 15*cm*	Jolly (1965), Manley-Parr and Fisher-Ford (Begon 1979)	Other mark and recapture models considered. Marks indicating week of mark and recapture applied during 1980-86.
1981	Mar 2-May 3	Freeze-brand	≥ 15*cm*	Jolly (1965)	
1982	Feb 22-May 11	Freeze-brand	≥ 15*cm*	Jolly (1965)	All winter flounder studies evaluated.
1983	Feb 21-Apr 6	Freeze-brand	≥ 20*cm*	Jolly (1965)	Minimum size increased so mostly adults marked. Survey limited to spawning season. Tow distance standardized. All marked fish sexed and measured.
1984	Feb 14-Apr 4	Freeze-brand	> 20*cm*	Jolly (1965)	
1985	Feb 23-Apr 2	Freeze-brand	> 20*cm*	Jolly (1965)	Jolly model evaluated.
1986	Feb 24-Apr 8	Freeze-brand	≥ 20*cm*	Jolly (1965)	

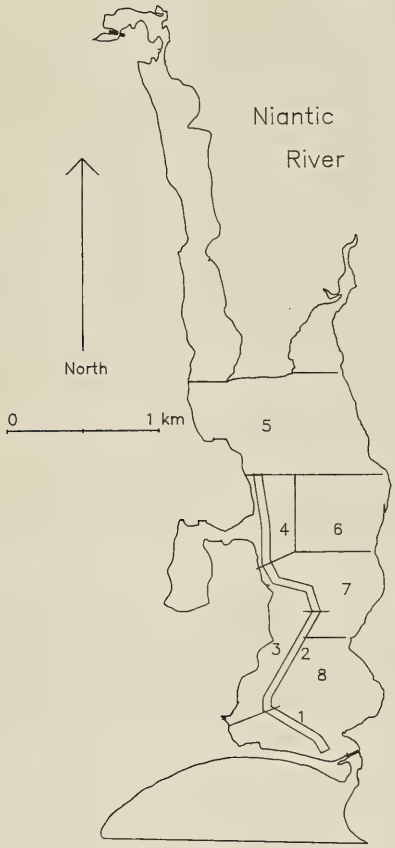


Figure 1. Location of Niantic River adult winter flounder sampling stations from 1976 through 1982.

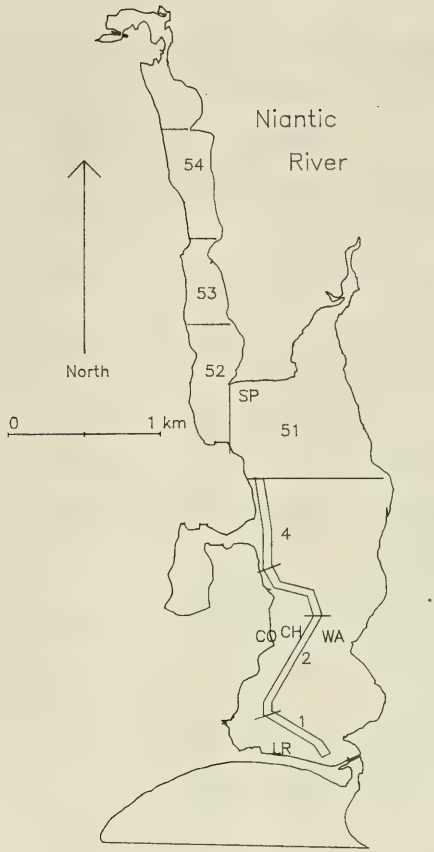


Figure 2. Location of Niantic River adult and juvenile winter flounder sampling stations from 1983 through 1986.

Winter flounder were captured with a 9.1-m otter trawl (6.4-mm bar mesh codend liner) towed by various vessels. From 1983 through 1986, tow was standardized at 0.55 km. This distance was chosen because it represented the maximum tow length at station 1 and because using the same tow length at all

stations was expected to reduce variability in calculating catch-per-unit-effort (CPUE), used as an index of abundance. However, because catch data from station 2 were also used for the trawl monitoring program, these hauls were maintained at a tow distance of 0.69 km. Prior to 1983, tows were not standardized. Mostly because of differences in tidal currents, wind, and amounts of extraneous material collected in the trawl, tow times for the standardized distances varied and were usually greater in the lower than in the upper river. For 1976-86, the mean duration for tows at stations 1 and 2 was 14.4 min and at stations 4, 51, 52, and 53 was 12.1 min. Tows from 1976-82 that had extremely short or long durations compared to the distribution of tow times from 1983-86, when tow distance was uniform, were excluded from data analyses and calculation of CPUE. For comparisons among years, all catches of winter flounder larger than 15 cm made during a 4-wk period from mid-March through early April were standardized to either 15-min tows (stations 1 and 2) or 12-min tows (all other stations). The annual mean and median CPUE were determined and a 95% confidence interval was calculated for each median using a distribution-free method (Snedecor and Cochran 1967). The catch of winter flounder taken in the trawl monitoring program from October 1976 through September 1985 (see Fish Ecology section for methods) was also used to calculate median CPUE values as indices of abundance for various size groups.

The winter flounder caught in each tow during the adult abundance survey in the Niantic River were briefly held in water-filled containers. At least 200 randomly selected fish were measured to the nearest 0.1 cm in total length during each week of the population abundance survey in all years. During 1983-86, all winter flounder larger than 20 cm were measured and sexed. Non-measured fish were classified into various length and sex groupings, depending upon the year; at minimum all fish caught can be classified as smaller or larger than 15 cm. Since 1977, the sex and reproductive condition of the larger winter flounder were determined either by observing eggs or milt or by the presence (males) or absence (females) of ctenii on the caudal peduncle scales of the left side (Smigielski 1975). Following measurement or classification, all fish 15 (1977-82) or 20 (1983-86) cm or larger were marked with a number or letter made by a brass brand cooled in liquid nitrogen; the mark was changed weekly. Fish recaptured were noted and remarked with the brand designating the week of their recapture. In 1976, fish were fin clipped in various ways and recaptures were marked with a numbered spaghetti tag.

Estimates of abundance of all winter flounder 15 or 20 cm and larger in the Niantic River during the spawning season were obtained from the mark and recapture data using the Jolly (1965) model. The actual computations were done using a computer program (Davies 1971) of Jolly's model with minor

modifications as described in NUSCo (1982). Prior to 1985, absolute estimates of winter flounder abundance during the spawning period had been obtained by starting with an estimate of N obtained during the first week of the survey and then adding the total number of fish joining (Jolly's B) during subsequent weeks. As a result of a comprehensive review of the mark-recapture methodology in 1985 (NUSCo 1986a), this procedure was eliminated. In its place a composite index was developed for describing the relative abundance of adult winter flounder. This index was computed by averaging the weekly estimates of N made during the winter flounder spawning season, except for the first and last estimates. These estimates are less reliable and were eliminated from the computations in all years except when the number of values used would have been less than three. The standard error of N was determined as:

$$(\tfrac{1}{3})\sqrt{\text{Var of}(N_2) + \text{Var of}(N_3) + \text{Var of}(N_4)} \qquad (1)$$

where N is the estimate of population size during each week

Fluctuations in the log-transformed catches of winter flounder taken at all six stations of the trawl monitoring program (see Fish Ecology section) were analyzed using harmonic regression techniques, methods of which were most recently described in NUSCo (1986a). Catch data from three replicated tows taken every other week were averaged to obtain a single biweekly mean. These values were used to construct various models describing catches from October 1976 through September 1985.

Life history studies

Various life history data have been collected since 1973 (Table 2). General methods and procedures for most studies briefly follow. For additional details concerning particular studies, specific reports should be consulted.

Reproduction

Using data from 1981-86, probit analysis (SAS Institute Inc. 1985) was used to estimate the length at which 50% of all females were mature. An index of the number of females reproducing in the Niantic River each year since 1977 was created by estimating their abundance in each 1-cm length increment starting with 26 cm. Fecundity (annual egg production per female) of Niantic River winter flounder was

Table 2. Summary of Niantic River adult winter flounder life history information collected from 1973 through 1986.

Year	Types of studies	Comments
1973	Length frequency, food habits, movements and exploitation.	Anchor tag returns used for movements and exploitation study
1974	Length frequency, food habits, movements and exploitation.	
1975	Length frequency	Not all adults measured from 1975 through 1983
1976	Length frequency, sex ratio	Not all adults sexed from 1976 through 1983
1977	Aging, length frequency, fecundity, length-weight, sex ratio and maturity	Both scale and otolith samples examined for age during 1977 and 1978
1978	Aging, length frequency, length-weight, sex ratio and maturity	
1979	Aging, length frequency, sex ratio and maturity, survival	Only scales used for aging from 1979 through 1983
1980	Aging, length frequency, sex ratio and maturity, survival, stock identification	Stock identification from isoelectric focusing of eye lens proteins by URI during 1980 and 1981
1981	Aging, length frequency, sex ratio and maturity, survival, stock identification, movements and exploitation	Petersen disc tag returns used for movements and exploitation studies from 1981 through 1983
1982	Aging, length frequency, sex ratio and maturity, survival, movements and exploitation	Life history studies evaluated
1983	Aging, length frequency, sex ratio and maturity, survival, movements and exploitation	All adults measured and sexed from 1983 through 1986. The von Bertalanffy growth model applied to data.
1984	Length frequency, sex ratio and maturity, movements and exploitation	Adult life history studies decreased in favor of increased larval and juvenile work.
1985	Length frequency, sex ratio and maturity	
1986	Length frequency, sex ratio and maturity	

estimated from length-frequencies and a length-fecundity relationship determined from 1977 data using a functional regression model with log-transformed variables (Jolicoeur 1975; Sprent and Dolby 1980). Forty-eight fish from 24.5 to 43.3 cm were examined according to methods found in NUSCo (1978a).

Data from an unpublished independent assessment of fecundity, which used 65 fish (19.7-45.3 cm) taken in Niantic Bay and at MNPS in 1977, were also used in a functional regression for comparative purposes. Annual mean fecundity was determined from the sum of all individual fecundities divided by the number of spawning females. The sum of the fecundities gave a relative annual index of egg production.

Age and growth

Separate length-weight relationships for Niantic River (270 specimens; 4.5-43.3 cm) and Bay (491; 6.7-45.3 cm) winter flounder were described with a functional regression with log-transformed variables. Specimens collected in 1977 and 1978 were measured to the nearest 0.1 cm in length and the nearest 0.1 g in weight.

During 1977-1982, randomly selected specimens were aged by examination of scales removed from the right side between the dorsal fin and the lateral line. Five or more scales from each specimen were cleaned and mounted in plastic resin on a slide and examined using a Bausch and Lomb trisimplex projector or a compound microscope. Except for the first year of life, winter flounder have a zone of widely-spaced circuli (fast spring and summer growth) followed by a zone of closely-spaced circuli (slow fall and winter growth). The outer edge of the closely-spaced circuli was considered to be an annulus (Lux and Nichy 1969; Lux 1973). Age of each specimen was determined by at least two people. Some comparisons of age were made during 1977-78 using otoliths with methods described in Williams and Bedford (1974) and Kurtz (1975). Annual age-length keys were constructed by determining the percentage that each of the ages made up of every 1-cm length increment in the sample of aged fish. This key was used to assign an age to all fish measured during the abundance surveys.

The growth rate of Niantic River winter flounder was determined by additional examination of one of the scales used in age determination in 1983. Based on previous findings, a stratified sample (Ketchen 1950; Ricker 1975) was used to select fish for aging. From five to ten scale samples were allocated to each 1-cm size interval of both sexes starting with 20 cm; scales from a number of smaller fish were also selected. Measurements were taken from the midpoint of the scale focus to each annulus and to the anterior margin of the projected scale image along a standard axis (Tesch 1968; Everhart et al. 1975). For the back-calculation of length-at-age, the relationship between scale size and fish length was examined. Some curvilinearity was seen in this relationship, especially for larger specimens, indicating probable

heterogeneous growth of the scale and fish (NUSCo 1984). Therefore, length at each annulus was calculated for each sex by the non-linear relationships:

$$\text{length} = 3.557(\text{scale size})^{0.908} \text{ for females, } (n = 216,\ r^2 = 0.93) \quad (2)$$

$$\text{length} = 3.777(\text{scale size})^{0.904} \text{ for males, } (n = 193,\ r^2 = 0.94) \quad (3)$$

Annuli measurements for each fish were substituted into the appropriate regression equation for back-calculation of growth. Mean lengths-at-age with 95% confidence intervals were then computed.

Using the 1983 length-at-age data, the von Bertalanffy growth model (Ricker 1975; Gallucci and Quinn 1979) was used to describe the growth of Niantic River winter flounder:

$$L_t = L_\infty \left(1 - e^{[-K\{t - t_0\}]}\right) \quad (4)$$

where L_t = length in mm at time t
K = growth coefficient
L_∞ = asymptotic maximum length
t_0 = hypothetical date at which a fish would have zero length if it had always grown in the manner described by the equation

A nonlinear procedure using the modified Gauss-Newton iterative method (SAS Institute Inc. 1985) was used to estimate the growth model parameters from the length-at-age data. The ω parameter (the product of L_∞ and K) of Gallucci and Quinn (1979) was calculated for comparisons of growth.

Similarly, the growth model was applied to the entire 1977-83 age-length data set. As all age 1 and 2 and some age 3 fish were not sexed, these specimens were used with both females (through age 10) and males (age 8). An independent assessment of growth was made using lengths at marking and recapture of 129 females and 81 males tagged with Petersen discs (see Movements and Exploitation below). As recommended by Sundberg (1984), the method of Fabens (1965) was used with these data to estimate L_∞ and K. Most recaptures used in the analysis were from NUSCo sampling to ensure that length data were

accurate. A few commercial returns and those from the Connecticut Department of Environmental Protection (CT DEP) and research institutions were also included. The data were constrained to include only fish captured after at least 90 d at large that showed positive growth; negative or zero growth was assumed to be due to measurement error or to severe effects of tagging which retarded growth.

Mortality and survival

Two methods were used to estimate survival (S) and the instantaneous mortality coefficient Z (= -ln S). The annual age-length keys from 1978-79 and 1981-83 were used with the length-frequency distributions of all measured fish 15 cm and greater (0.5-cm groupings) to determine total number by age. Catch curves were constructed and the slope of the natural logarithm of number plotted against age was used as an estimate of Z (Ricker 1975). Estimates of survival were also made using the method of Robson and Chapman (Robson and Chapman 1961; Ricker 1975):

$$S = \frac{T}{\sum N + (T - 1)} \qquad (5)$$

where $T = N_1 + 2N_2 + 3N_3 + \ldots$

$\sum N = N_0 + N_1 + N_2 + \ldots$

N_0 = number of age 3 winter flounder

N_1 = number of age 4 winter flounder ...

$$\text{variance} = S\left(S - \frac{(T - 1)}{\sum N + (T - 2)}\right)$$

Food habits

The food items of 306 winter flounder collected in the area of Millstone Point from June 1973 through November 1974 were examined. Whole stomachs were removed and preserved in 10% buffered formalin. Each stomach was cut open, examined, and subjectively ranked according to fullness, and assigned point values (Hynes 1950) as follows:

100 - full; appeared unable to hold any additional ingested material

75 - three-quarters full; stomach distended such that no folds seen

50 - half-full; partial stomach distention

25 - one-quarter full; some ingested material, but little distention

0 - empty; no ingested material

The stomach contents were sorted and classified to a major taxonomic grouping. A visual estimate was made of the percentage of stomach contents for each taxon. The fullness point total for each stomach was multiplied by the percentage of each taxon to give a point value for each food item and averages were computed for each station.

Movements and exploitation

During February through July of 1973 and 1974, approximately 2,000 and 2,600 winter flounder, respectively, were individually marked with a Floy anchor tag and released in various areas near Millstone Point. The tagging study was performed by a consultant and initial tagging data are not available which limits the usefulness of the study. However, tag returns that were reported in NUSCo (1975) may be used to make inferences concerning movements and exploitation by the sport and commercial fisheries.

From December 1980 through September 1983, almost 5,000 specimens larger than 20 cm were marked with a Petersen disc tag. During tagging operations, length and sex information were recorded along with location and date of release. A white 1.3-cm diameter disc uniquely numbered and printed with information for its return was positioned on the nape of the right side of the fish and a red disc with additional information was used on the left side. A nickel pin was pushed through the musculature, cut to size, and its end was crimped over to connect the tags and hold them in place. Except for specimens released specifically at the MNPS intakes, winter flounder were returned to the location of their capture. Information requested at recapture included date, location, method of capture, length, sex, and additional scales. A reward of $1.00 was given to all persons returning a tag.

Stock identification

A special study was undertaken for NU by the University of Rhode Island to determine if local populations of winter flounder could be distinguished using a biochemical technique, direct tissue isoelectric

focusing. Theory and details regarding this electrophoretic method may be fround in the original report (Schenck and Saila 1982) as well as in Lundstrom (1977), Saravis and Zameheck (1979), and Marine Colloids (1980). Samples were taken during the spawning season when stock separation should have been greatest. From February through April 1980, approximately 50 winter flounder were collected from New Haven, CT; Connecticut River; Niantic River, Niantic Bay, and Jordan Cove (combined for analysis); Thames River, CT; Mystic River, CT; and Charlestown Pond, RI. In March 1981, about 75 fish were examined from the Connecticut River, Niantic River, Niantic Bay, Jordan Cove, and Thames River. Additional samples were collected in Niantic Bay during each season to examine changes on an annual basis. All except the New Haven fish were processed immediately after collection. Each fish was pithed, measured, weighed, sexed and its eye lenses were removed and individually frozen. After the eye lens proteins were separated, sample gels for each specimen with separated protein bands were fixed and stained and examined for the presence or absence of certain protein bands. Some samples were quantitatively read with a scanning densitometer. Data were analyzed using linear discriminant analysis, details of which may be found in Schenck and Saila (1982).

Larval studies

Abundance and distribution

Ichthyoplankton collections containing larval winter flounder have been made at numerous locations in the Millstone area since 1973. Prior to 1979, the collection of winter flounder was incidental to the general ichthyoplankton surveys, with varying objectives. Since 1979, special sampling in the Niantic River has been conducted during the occurrence of larval winter flounder. The most comprehensive sampling has been conducted since 1983, with numerous special studies conducted to identify sampling biases. Except for entrainment sampling (discussed below), the varying sampling designs prior to 1983 have limited the usefulness of these data in understanding factors affecting the abundance of larval winter flounder in the Millstone area.

All offshore ichthyoplankton sampling designs have used 60-cm bongo samplers towed from boats at approximately 2 knots and weighted with various depressors of differing weights. Sample volumes were determined with General Oceanic flowmeters (Model 2030). Mesh size of the nets have varied with paired

333- and 505-μm mesh from 1973 through 1978, exclusively 333-μm mesh from 1979 through 1983, and varying combinations of 202- and 333-μm nets in 1984 and 1985. Tow duration was usually 15 min (volume filtered of about 200-250 m^3) through 1984, but was reduced to 6 min (ca. 100-150 m^3) in 1985 for Niantic River collections. Most tows were oblique, but the tow pattern was changed in 1980 from continuous (bongo sampler continuously retrieved or let out between surface to near the bottom) to stepwise (equal sampling time at surface, mid-depth, and near the bottom). In addition, surface and bottom tows were taken from 1973 through 1978. Bottom tows were taken with a bongo sampler mounted on a sled. All samples were preserved in 5 to 10% formalin.

Offshore sampling programs can be categorized into four sampling periods. An intensive area-wide sampling took place from May 1973 through 1975. A much reduced area-wide sampling occurred from 1976 through 1978. Sampling efforts increased for winter flounder larvae in the Niantic River and at one station in Niantic Bay from 1979 through 1982. Finally, more comprehensive sampling took place in the Niantic River from 1983 through 1985.

The intensive area-wide sampling program began in May of 1973 to verify the winter flounder larval dispersal model (Sissenwine et al. 1973); a detailed sampling scheme was provided in NUSCo (1976). Because sampling started in May 1973, information on larval winter flounder was restricted to 1974 and 1975. Stations 1-13 were sampled in 1974 and 1-16 in 1975 (Fig. 3). During this period, sampling frequency varied, but usually an oblique tow was made at each station during the day once a week and at night once every month. Surface and bottom tows were taken every other week during the day and night at selected stations. For this report, these data were examined for temporal distribution of winter flounder larvae in the Millstone area and for the collection efficiency of 333- and 505-μm mesh nets.

In 1976, the ichthyoplankton program was reduced because the previous data were not adequate for verification of the winter flounder larval dispersion model (Vaughan et al. 1976). The number of stations was reduced to six (2, 5, 6, 8, 11, and 14). Single day oblique tows were made monthly with additional surface and bottom tows in May through August. Because of the low frequency of sampling (monthly) at each station, these data were not useful to examine the life history of larval winter flounder in the Millstone area.

Figure 3. Location of stations sampled for larval winter flounder at various times from 1976 through 1985.

Based on a 1978 evaluation of the offshore ichthyoplankton program (NUSCo 1978b), stations were further reduced to one (5) with an increase in sampling frequency in 1979. From January through March and September through December, one day and night oblique tow was taken every other week. During the remaining months, two day and night tows were made weekly, with all replicate samples processed. This sampling program has remained the same to the present. For reporting purposes, the station 5 designation was changed to NB in 1983.

Specific sampling for larval winter flounder during their occurrence in the Niantic River began in 1979. Some changes have been made in sampling frequency, station location, and station designation from 1979 through 1985 (Table 3). The Niantic River sampling was re-designed for 1983, based on the recommendations of a larval winter flounder workshop held at NUEL in October 1982. The changes included special studies to reduce sampling biases, resulting from tidal and diel larval behavior. Identification of sampling biases with special studies in 1983 (NUSCo 1984) and 1984 (NUSCo 1985) limited the usefulness of data collected previously in 1979-82.

Starting in 1983, ichthyoplankton samples for winter flounder larvae were taken in Niantic River at stations A, B, and C and in mid-Niantic Bay at station NB (Fig. 3). To ensure mid-depth and near bottom sampling, the length of tow line necessary to sample the mid-water and bottom strata was based on water depth and the tow line angle measured with an inclinometer (NUSCo 1984). Nets with 333-µm mesh were used in 1983. Paired 202- and 333-µm mesh nets were towed in 1984 through mid-March and only 333-µm mesh nets during the remainder of the season. In 1985, 202-µm mesh nets were used through the first week of April and 333-µm mesh nets during the remainder of the season. Medusae of a potential predator, the lion's mane jellyfish (*Cyanea* sp.), were sieved (1-cm) from the samples at the three river stations and measured volumetrically (ml). These data were compared to mean weekly medusoid diameter in the river (Miller et al. 1986) and a diameter to volume relationship was used to estimate abundance of medusae.

Sampling time and frequency varied during the 1983-85 surveys. In 1983, sampling time in the Niantic River was systematically varied during daylight and night and at station C sampling was varied over four tidal stages (high, low, mid-ebb, and mid-flood). At each of the three river stations, day and night tows

Table 3. Summary of sampling in the Niantic River for larval winter flounder.

Year	Sampling frequency	Replicates	Stations
1979	day - weekly	2	2
1980	day and night - weekly	4	2
1981	day and night - weekly	4	1,2
1982	day and night - weekly	2	1,2
1983	day and night - twice weekly	1	A, B, C
1984	day and night - twice weekly[a]	1	A, B, C
1985	day and night - twice weekly[a]	1	A, B, C

[a] Day samples only through the first week of April; day and night samples during the remainder of April; night samples only during the remainder of the season.

were made twice weekly (Monday and Thursday or Tuesday and Friday). In 1984 and 1985, from the beginning of sampling (mid to late February) through the first week in April, single tows were made during the day twice weekly within 1 h of low slack tide. During the last 3 wk of April, single-bongo tows were made twice weekly day and night. The day samples were collected within 1 h of low slack tide and the night samples during the second half of a flood tide. During the remainder of the season until the disappearance of larvae at each station, tows were made twice a week only at night during the second half of a flood tide.

Laboratory sample processing has varied (Table 4). Through 1977, whole samples were processed and all winter flounder larvae were measured. Subsequently, samples have been split based on the abundance of larvae and up to 50 winter flounder larvae were measured. When a subsample of larvae was measured, the length-frequency distribution was adjusted by sample density. Prior to 1983, measurements were in total length and since then have been in standard length (tip of snout to end of notochord). The difference between total and standard length measurements were only apparent for larvae in the latter stage of metamorphosis when the caudal fin rays extended past the end of the notochord; these larger larvae were collected infrequently. From 1973 through 1976, both paired 333- and 505-μm samples were processed, but length measurements taken only in 505-μm samples in 1976. In 1977 and 1978, only the 333-μm samples from the paired 333- and 505-μm mesh nets were processed. From 1979 through 1982,

both replicated 333-μm samples from each bongo tow were processed. Since 1983, only one of the two bongo sampler replicates collected in the Niantic River was processed in the laboratory and the developmental stage of each measured larva recorded. The five stages were defined as:

Stage 1. The yolk sac was present or the eyes were not pigmented (yolk-sac larvae)

Stage 2. The eyes were pigmented, no yolk sac was present, and no fin ray development

Stage 3. Fin rays were present, but the left eye had not migrated to the mid-line

Stage 4. The left eye had reached the mid-line, but juvenile characteristics were not present

Stage 5. Transformation to juvenile was complete and intense pigmentation was present near the caudal fin base

Because few Stage 5 larvae were collected, their abundance and distribution were not examined.

Table 4. Summary of mesh size and laboratory processing of larval winter flounder samples.

Year	Sampling mesh size (μm)	Sample splitting	No. larvae measured
1974-76	333,505	no	all
1977	333	no	all
1978-83	333	yes	up to 50
1984-85	202,333	yes	up to 50

Larval data analyses were based on standardized densities per 500 m^3 of water sampled. Weekly mean densities were used because of varying sampling frequencies. For comparisons of data in the 1981-85 period, daylight samples during 1981-83 from the last week of April through the end of the season were

excluded. During these weeks in 1984 and 1985, daylight samples were not collected because these samples underestimated abundance due to diel behavior of the older larvae, which apparently remained near bottom during the day and were not susceptible to the bongo sampler (NUSCo 1984).

Typically, the distribution of larval abundance over time is skewed, with a rapid increase to a maximum followed by a slower decline. This skewed density distribution results in a sigmoid-shaped cumulative distribution and the time of peak abundance is the time at which the inflection point occurs in the cumulative distribution. The cumulative Gompertz function (Draper and Smith 1981) was chosen to describe the cumulative distribution data because the inflection point of the Gompertz function is not constrained to the central point of the sigmoid curve. The form of the cumulative Gompertz function used was:

$$C_t = \alpha\left(\exp[-\beta e^{-kt}]\right) \tag{6}$$

where C_t = cumulative density at time t
α = total or asymptotic cumulative density
β = location parameter
k = shape parameter
t = time in days from February 15

The origin of the time scale for our data was set to the 15th of February, which is when winter flounder larvae generally appear in the Niantic River. The parameter α was used as an index to compare annual abundances.

The derivative of the above cumulative function with respect to time yields a "density" function which directly describes the larval abundance over time. This density function has the form:

$$d_t = \alpha\beta k\left(\exp[-kt\{-\beta e^{-kt}\}]\right) \tag{7}$$

where d_t = density at time t

where all the parameters are the same as in the cumulative function (Equation 6), except for α, which was

rescaled by a factor of 7 because the cumulative densities were based on weekly means and thus accounted for a 7-d period. Time of peak abundance was estimated as the date t_i corresponding to the inflection point of the cumulative function defined by its parameters β and k as:

$$t_i = \frac{(\ln \beta)}{k} \tag{8}$$

Least-squares estimates and asymptotic 95% confidence intervals for these parameters were obtained by fitting Equation 6 to the cumulative abundance data using nonlinear regression methods (SAS Institute 1985).

The cumulative Gompertz function was used to examine temporal and spatial distribution of larval winter flounder from data collected in the 1974-75 offshore study, entrainment sampling in 1976-85, and special larval winter flounder sampling in 1981-85. The stations sampled during 1974 and 1975 were grouped in relation to their location to the Niantic River: Niantic River (1 and 2); mouth of the Niantic River (3, 4, 15, and 16); mid-Niantic Bay (5, 11, and 12); Jordan Cove (6 and 13); Twotree Island Channel (7 and 8); and Offshore (9, 10, and 14).

Time-based harmonic regression models were also used with abundance data from station 5 in Niantic Bay (1979-85) and the entrainment sampling station EN (1976-85) to describe fluctuations in abundance (see Fish Ecology section for detailed methods).

Special studies

The only station in the Niantic River with strong tidal currents was C. To determine the effect of tidal currents on sample densities of larvae, 24-h studies were conducted in 1983 and 1984. Samples were collected at 2-h intervals over a 24-h period on April 28 and May 9 in 1983 and March 12 and March 19 in 1984. Tow duration was 15 min with paired 333-µm mesh nets in 1983 and 6 min with paired 202- and 333-µm mesh nets in 1984. These data were examined to determine if tidal currents caused sample density biases.

Larval import and export studies were conducted at the mouth of the of the Niantic River from 1983 through 1985. Stationary tows were taken by mooring the boat to the Niantic River Highway Bridge in the middle of the channel. Bongo samplers were deployed off each side of the boat with one at mid-water

and the other near bottom. In 1983, samples were collected at the time of maximum ebb and flood tidal currents during three tidal cycles (two cycles on May 9 and one on May 16). In 1984 and 1985, samples were collected hourly except for 1 h before and after slack tidal currents during five tidal cycles (April 4 and May 8 in 1984; March 28, April 29, and May 28 in 1985). The 1984 and 1985 data were combined along with current velocity from the flowmeters to calculate the net exchange of larvae leaving and entering the river.

Ebb and flood tide velocity measurements used in estimating net larval exchange may not have been comparable due to the different widths of the channel at the point of sampling. Due to the length of the mooring line tied to the bridge, the actual sampling location was approximatedly 10 m north of the bridge during a flood tide and approximately 10 m south of the bridge during an ebb tide. The comparability of velocities was investigated by fitting a second order polynomial equation to the water velocity measurements over time during the five flood and ebb tidal phases sampled in 1984 and 1985. Because there is minimal freshwater input into the Niantic River, the area under the fitted curves for flood and ebb tides should be similar in magnitude. If the areas differ, then adjustments to velocity measurements of either tidal stage could be made to make the areas similar and velocities comparable (NUSCo 1986a).

The effects of mesh size and tow duration on net extrusion of larval winter flounder were examined. Comparisons of mesh size were examined in the laboratory (NUSCo 1986a) and in field studies (NUSCo 1985). In the laboratory, meshes of 202, 333, and 505 μm were compared. An apparatus was constructed such that the velocity of water could be regulated as it passed through a chamber covered by the various meshes with a similar cross-net velocity (ca. 20 cm/sec) as encountered in field sampling. Ten laboratory-reared yolk-sac to first-feeding larvae (ca. 3-4 mm) were placed in the chamber. The flow was maintained for 15 min, the chamber was removed, and the number of larvae retained was counted. From 9 to 18 tests were completed for each mesh. Field comparisons of 333- and 505-μm mesh nets were based on 492 bongo tows made in 1974 and 1975, which examined the number of larval winter flounder by 1-mm size classes. Comparisons of 202- and 333-μm mesh nets were available from 28 bongo tows in 1984 and were based on the sample densities of Stage 1 and early Stage 2 larvae. The effect of tow duration on sample density was made in 1984 with consecutive 6- and 15-min tows in the Niantic River using 333-μm mesh nets (16 comparisons). A Wilcoxon signed-ranks test was used to compare the paired samples and test for significant differences ($p \leq 0.05$) due to mesh and tow duration.

Otoliths from larval winter flounder collected in the Niantic River during 1984 were examined to determine if an age-length key could be constructed from daily increments (NUSCo 1985). At least one sample each week from all Niantic River stations was preserved with 95% ethanol and processed to obtain approximately 50 larvae, if possible. All otoliths removed (1 to 4) from a larva were mounted and examined with a compound microscope connected to a video monitor that provided a magnification of approximately 5,000 X. If individual otoliths differed in size, the larger pair was assumed to be the sagitta and used for counting increments. If no size difference was found, the otolith with the most distinct increments was used.

Winter flounder larvae were reared in the laboratory during 1985 to determine developmental time and to verify daily otolith deposition (NUSCo 1986a). To examine the effect of starvation on growth, larvae in one aquarium were not fed. Known-age larvae were routinely sacrificed to obtain otoliths for aging verification and information on growth rate. Sampling frequency varied, with almost daily collections during early development to approximately biweekly during later development when few larvae remained.

Post-larval studies

Abundance and distribution

The CPUE of juveniles smaller than 15 cm taken in the Niantic River during the spawning season was determined in a manner similar to that for adults. Their abundance at the trawl monitoring program stations was also examined. Numbers of these fish, most of which were age 1, provided information on the relative strength of year-classes produced in local waters. In addition, during the surveys of 1981 and 1982, 6- to 15-cm juveniles were marked with freeze-brands to obtain abundance estimates using the Jolly model.

The abundance and distribution of young-of-the-year (age 0) winter flounder were first examined during 1976 through 1978. Brief field studies were conducted in summer using various seines and trawls and diver observations in the Niantic River (Table 5). Because of the small effort and difficulties in quantifying diver observations without bias, these studies did not provide useful information and will not be discussed further.

Table 5. Summary of Niantic River post-larval juvenile winter flounder studies from 1976 through 1985.[a]

Year	Method	Period sampled	Areas or stations sampled	Times sampled	Comments
1976	otter trawl	May 27-Jul 23	4-8	3	Confirmed Niantic River
	seines	Jun 16	5	1	as nursery area
1977	otter trawl	May 17-Jul 29	3	5	Inconclusive - few taken
	diving obs.	Aug 9-Sep 9	12	2	Rarely seen except in lower river
1978	diving obs	Jul - Sep	11	3	Abundance estimates made based on observed densities and bottom areas
1981	freeze-branding	Apr - May	4	--	6 - 15 cm juveniles branded during part (1981) or
1982	freeze branding	Feb - May	4	--	all (1982) of adult surveys. Abundance estimates unreliable because of high marking mortalities
1983	beam trawl	May 18-Oct 12	4	21	Abundance, growth and mortality estimates made from 1983 through 1985
1984	beam trawl	May 24-Sep 26	3	19	A seine was also used several times in 1983, but few were taken.
1985	beam trawl	May 23-Sep 19	2	18	

[a] Juvenile ($\leq$ 15 cm) winter flounder abundance estimated by CPUE also available for 1976 through 1985 from adult winter flounder surveys in Niantic River and from trawl monitoring program data.

A quantitative study of post-larval young-of-the-year winter flounder in the Niantic River began in 1983 and has continued through the present. One of the four stations established then (LR) was sampled in each year (Fig. 2). Station CO was sampled from 1983 through July 1984, when it was replaced by WA because continued heavy accumulations of the alga *Enteromorpha clathrata* hampered sampling at the former location. Stations SP and CH were used only in 1983 and were dropped because flounder were less common there and catches more variable; data from these two stations will not be included in this report. All stations contained habitat preferred by juvenile winter flounder, with sandy to muddy bottoms

in shallow (ca. 1-2 m) water adjacent to eelgrass beds (Bigelow and Schroeder 1953). Each was sampled once every week from late May through late September or early October during daylight within about 2 h before to 1 h after high tide.

A 1-m beam trawl was used with interchangeable nets of 0.8-, 1.6-, 3.2-, and 6.4-mm bar mesh. A tickler chain was added in late June of 1983 to increase catch efficiency. In 1983, triplicate tows were made using one of the nets, which was changed as young grew during the season. In 1984 and 1985, two nets of successively larger mesh were used during each sampling trip to collect the entire available size range of young. This helped to eliminate a bias that was found in 1983, when some of the older and larger specimens apparently were able to avoid the 0.8-mm mesh net used without a tickler chain (NUSCo 1984). A change to the next larger mesh in the four-net sequence was made when young had grown enough to become susceptible to it. The larger meshes also reduced the amount of detritus and algae retained. Two replicates with each of the two nets were made at both stations; the order in which the nets were deployed was chosen randomly. Distance of each tow was estimated by letting out a measured line attached to a lead weight as the net was towed. Tow length increased from 50 to 75 to 100 m as the number of fish decreased throughout the summer. For data analysis and calculation of CPUE, the catch of both nets used at each station was summed and standardized to give a density per 100 m^2 of bottom covered by the beam trawl.

Growth and mortality

Young winter flounder were measured unpreserved in the field or laboratory to the nearest 0.5 mm in total length (TL). During the first few weeks of study, standard length (SL) was also measured because many of the specimens had damaged caudal fin rays and total length could not be taken. The relationship between the two lengths was determined by a functional regression and used to convert SL to TL:

$$TL = -0.10 + 1.198\ (SL)\ (n = 136,\ r^2 = 0.92) \qquad (9)$$

To calculate mortality, all young were assumed to comprise a single cohort. A catch curve was constructed with the natural logarithm of density plotted against week. The slope of the descending portion of the curve provided an estimate of Z, the weekly rate of instantaneous mortality. Once Z was determined, daily survival was estimated as exp(-Z/7), weekly as exp(-Z), and monthly as exp((-Z)(30.4/7)).

Impingement

The number of winter flounder impinged on the traveling screens of MNPS from October 1972 through September 1985 was estimated using techniques described in the Fish Ecology section of this report. The chronology of impingement sampling at MNPS was also given there. Meteorological data were obtained from MNPS operating records to examine certain specific instances of high impingement. Length-frequency data of fish impinged from 1976-77 through 1984-85 were also examined. The sex and reproductive condition of impinged winter flounder taken just prior to and during the spawning 1982 and 1983 seasons were recorded using methods and criteria described previously.

Details regarding studies leading to the construction, installation, and operation of a fish return sluiceway at MNPS Unit 1 were summarized in the Fish Ecology section. Data and findings pertinent to the winter flounder will be discussed in this section. Additional information may be found in NUSCo (1981b, 1986b).

Entrainment

Winter flounder larvae entrained at MNPS were collected at station EN (Fig. 3). Sampling has been conducted since April 1973 as part of a general ichthyoplankton sampling entrainment program. Because of sampling problems, data collected through June 1975 were not used to estimate entrainment numbers, but were the basis for the sampling design beginning in July 1975. Entrainment samples from the 1976-85 larval winter flounder seasons (primarily March through May) provided the longest time-series of data for which annual comparisons could be made. The only changes in sampling design were sampling frequency and replication. For 1976-82, three replicates were collected day and night on 3 d each week. In 1983, sampling frequency was changed to one sample during the day and night on 4 d per week. Sampling alternated weekly at the discharge of Units 1 and 2, whenever plant operations permitted. Approximately 400 m^3 of water were filtered through a 1.0-m diameter, 3.6-m long, 333-μm mesh conical plankton net. Laboratory processing methods and some data analyses were described previously in this report. Additional details may be found in the Fish Ecology section, including the method used to estimate annual entrainment numbers.

An entrainment mortality study was conducted in 1983 (NUSCo 1984). Samples of entrained winter flounder larvae were collected downstream from the MNPS discharges with a 0.5-m diameter, 1-m long, 333-μm mesh plankton net. Dead larvae were removed from the sample and counted. Live larvae were held in flow-through chambers in effluent water. Holding time was 2 or 4 h to simulate retention in the quarry with three or two units in operation, respectively. Following effluent holding, chambers were moved to ambient flow-through water and observed daily for latent mortality over a 96-h period.

The thermal tolerance of larval winter flounder was examined (NUSCo 1975). Larvae were grouped as pre-metamorphosis (< 5 mm) and metamorphosing (≥ 5 mm). Larvae were exposed to a ΔT of 13 °C (acclimation temperature of 8 °C) for 1 to 9 h and survival was monitored. The critical thermal maximum for the two groups was also examined, where larvae held at 8 °C were exposed to an increase of 1 °C per min until death. The temperature at which complete mortality occurred was the estimated critical thermal maximum of larval winter flounder.

Impact assessment

A population dynamics model was developed under the direction of Dr. Saul Saila of the University of Rhode Island for impact assessment during the projected life of the plant and for a recovery period following decommissioning. This model incorporates hydrodynamic, concentration, and population submodels in a simulation of the effects of MNPS operations on the Niantic River stock of winter flounder. Data from both NU studies and the scientific literature were used in the model. The methods and procedures used in its development and may be found in progress reports to NUSCo by Sissenwine et al. (1973, 1974, 1975) and Vaughan et al. (1976). Aspects of the model were published by Hess et al. (1975). The final form of the model presented by Saila (1976) was summarized and used with updated information in the Environmental Report for MNPS Unit 3 (NUSCo 1983c). A brief summary of the results and conclusions of the latest model application will be given in this report.

RESULTS AND DISCUSSION

Adult abundance studies

Abundance in the Niantic River

The Niantic River winter flounder population is demographically open and therefore subject to immigration, emigration, natural death, and removal by fishermen. Although attempts to estimate the abundance of Niantic River winter flounder using mark and recaptures techniques began in 1973, the first reliable estimates were not made until 1976. The sampling intensity was too low and effort spread across too much area and time in 1973 and 1974 for estimates to be made by any means. In 1975, a more intensive sampling program was undertaken in the river. Fish were marked by fin clips, but recaptures were not marked. Nevertheless, the Jolly (1965) model, a multiple mark and recapture method, was applied to the data. Multiple recaptures were mathematically simulated, but the resulting estimates were judged to be inadequate because of large errors in parameter estimates. In addition, the 1975 survey data were not similar to later years for calculation of CPUE or other relative abundance indices. The March 31 start meant that the survey missed most of the spawning and was not temporally comparable to later years. Most tows were very brief (average of 5 min) and definition of stations differed from later years.

Surveys expressly designed to estimate abundance of open populations using the stochastic model of Jolly began in 1976. The Jolly model is an extremely powerful general formula that uses all the information provided by the mark and recapture experiment and provides the most efficient estimates as long as failures do not occur in basic assumptions (Cormack 1968; Southwood 1978; Begon 1979). The second measure of abundance for winter flounder over the past 11 yr was the CPUE during a 4-wk period from mid-March through early April, the only time including comparable data for all surveys. The median CPUE was used as the most appropriate catch statistic as the trawl catch data were not normally distributed and were positively skewed.

Annual Jolly composite abundance indies were calculated for 1976-86 using the mark and recapture data (Table 6). The 1976-82 surveys extended into May, but later ones were made only during the period when most spawning had occurred and ended by mid-April. Consequently, the earlier data were re-analyzed

to conform to the same criteria for ending surveys after 1982 (Appendices I - XXII). From 5 to 8 wk of mark and recapture data annually were available for the Jolly model. The largest number of fish branded in one year was in 1981 (6,726) and totals have since declined each year through 1986 (2,790). Fewest recaptures were made in 1985 (143; 4.7% of the total branded) and the most in 1980 (433; 10.0%); the average annual recapture rate was 6.7%.

Table 6. Yearly mark and recapture data for Niantic River winter flounder studies from 1976 through 1986.

Year	Dates sampled[a]	No. of weeks	No. marked[b]	No. recaptured	% recaptured
1976	Mar 1 - May 4	10	9,856	699	7.1
	Mar 1 - Apr 13	7	6,479	453	7.0
1977	Mar 7 - May 10	10	6,860	623	9.1
	Mar 7 - Apr 12	6	3,737	257	6.9
1978	Mar 6 - May 16	11	8,403	729	8.7
	Mar 6 - Apr 25	8	4,417	360	8.2
1979	Mar 12 - May 15	10	8,105	491	6.1
	Mar 12 - Apr 17	6	4,067	241	5.9
1980	Mar 17 - May 6	8	7,625	961	12.6
	Mar 17 - Apr 15	5	4,313	433	10.0
1981	Mar 2 - May 3	10	10,458	822	7.9
	Mar 2 - Apr 14	7	6,726	469	7.0
1982	Feb 22 - May 11	12	11,076	901	8.1
	Feb 22 - Apr 6	7	5,795	270	4.7
1983	Feb 21 - Apr 6	7	5,196	363	7.0
1984	Feb 14 - Apr 4	8	3,740	197	5.3
1985	Feb 27 - Apr 10	7	3,024	170	5.6
1986	Feb 24 - Apr 8	7	2,790	175	6.3

[a] For 1976-82, first line gives data for the entire survey and second for the spawning season.

[b] Minimum size for marking was 15 cm during 1976-82 and 20 cm thereafter.

The Jolly composite abundance indices showed that the Niantic River winter flounder population was relatively stable from 1976 through 1980, increased to a peak in 1982, and declined to an 11-yr low in 1986 (Table 7; Fig. 4). Using the catch of fish between 15.0 and 19.9 cm, the estimates for 1983-86 were also adjusted upwards for direct comparability between that period and 1976-82, as the estimates for the earlier years included all fish larger than 15 cm. The annual trends in median CPUE generally corresponded

with the Jolly composite index of abundance until 1982 (Table 8; Fig. 4). The CPUE in 1982 (42.6) was nearly the same as in 1981 (43.4), but the abundance index increased 72% (28,693 to 49,439). However, the Jolly estimate during 1982 was relatively imprecise with a large confidence interval (± 16,666). The decline in CPUE for fish larger than 20 cm from 1983 (22.1) to 1984 (12.8) and 1985 (12.6) was much greater than for the abundance index (29,912 to 29,282 and 21,632). The Jolly index for 1986 (8,252) declined greatly from 1985, but the CPUE decreased by only about 20%. The CPUE for 1984-86 indicated population levels about one-half of that during 1976-80, which also contradicted the Jolly abundance indices. The apparent differences between abundance indices were evaluated below by examining methodologies used in obtaining the estimates.

Data from the trawl monitoring program for the five stations outside the Niantic River were used in calculating a median CPUE during January through April of each year. This period overlapped the river spawning and allowed for an increase in available data. The median CPUE showed relatively low densities

Table 7. Composite index of abundance for Niantic River winter flounder from 1976 through 1986.

Year	No. of values used	Composite abundance ± 2 standard errors[a]	Adjusted abundance
1976	3	21,795 ± 4,768	---
1977	3 [b]	18,183 ± 5,088	---
1978	3 [c]	14,620 ± 3,494	---
1979	2 [c]	18,709	---
1980	3 [c]	18,159 ± 3,976	---
1981	3	28,693 ± 6,640	---
1982	3	49,439 ± 16,666	---
1983	3	29,912 ± 7,042	34,997
1984	3	29,282 ± 10,518	36,310
1985	3	21,632 ± 9,345	26,175
1986	3	8,252 ± 2,723	9,663

[a] For winter flounder larger than 15 cm during 1976-82 and 20 cm thereafter. Abundance adjusted to all fish larger than 15 cm for 1983-86.

[b] Only N_1 excluded.

[c] No values of N excluded.

of adult winter flounder outside the river in winter and early spring, with little change in the catch of adults over the 11-yr period. Values ranged from 1.8 to 5.3 in all years except 1981, when a peak value of 8.2 was found. Although the Niantic River spawning population apparently reached its lowest level in 1986, the median outside the river was the third highest (4.7). The reason for this difference in abundance and distribution is unknown. Water temperature is an important factor in winter flounder distribution. However, based on MNPS operating records, water temperature in 1981 was the closest to the annual February-April mean over the 11-yr period and the mean for 1986 was only slightly higher than 1985. The coldest year of the series was 1976 and the warmest was 1983.

Table 8. Mean and median CPUE of Niantic River winter flounder larger than 15 cm from 1976 through 1986 during the period of mid-March through mid-April.

	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986
Total tows made	112	154	106	93	112	109	90	135	145	156	179
Tows used for CPUE	85	123	88	77	91	97	87	134	143	155	173
% of tows used	76%	80%	83%	83%	81%	89%	97%	99%	99%	99%	97%
Mean CPUE	37.6	29.8	24.2	38.8	41.4	48.4	47.6	31.5	16.9	15.6	14.4
Standard deviation	33.5	25.8	16.0	33.6	32.3	31.1	30.6	16.3	10.2	8.0	9.0
Coeff. of variation	89%	86%	66%	87%	78%	64%	64%	52%	60%	51%	63%
Median CPUE	28	24	19.6	26.8	31.5	43.4	42.6	30.8	15.0	14.7	12.0
95% CI	22.5	20.0	16.2	22.4	26.1	36.2	35.2	24.1	13.6	12.7	10.6
	-37	-30	-25	-38.4	-42.5	-51.4	-48.8	-33.9	-16.6	-15	-14.6
Coeff. of skewness[a]	2.33	1.45	1.18	1.67	1.54	1.24	1.13	0.96	1.48	1.13	1.25

[a] Zero when data are distributed symmetrically

Both the Jolly model and its application to the Niantic River winter flounder were evaluated several times by NUEL (NUSCo 1980, 1983b, 1986a). In the first two reviews, data were examined for the violation of assumptions or conditions of the model generally as set forth by Bishop and Sheppard (1973), Roff (1973a, 1973b), Southwood (1978), Begon (1979), Balser (1981), and Arnason and Mills (1981). Most assumptions were met, although the possibility existed that some emigration outside the survey area was not permanent and marked winter flounder re-entered the river. This assumption requires that all emigration from the study area be permanent and therefore indistinguishable from death in the model. If

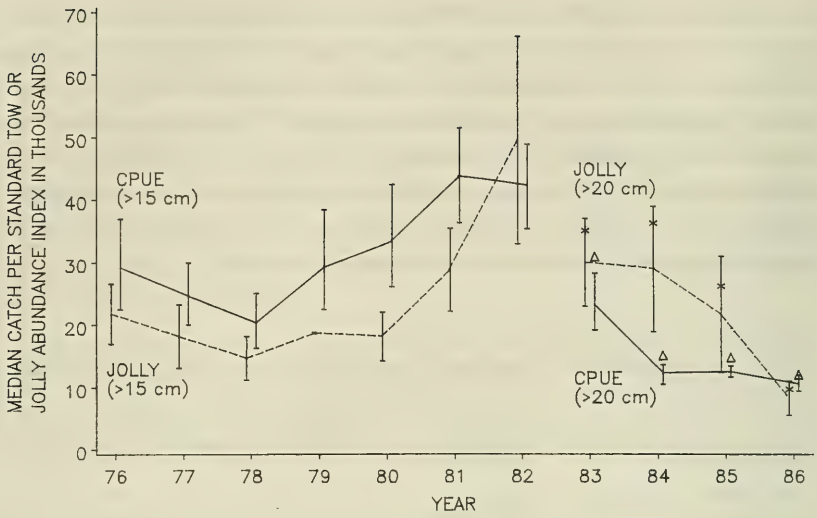


Figure 4. Annual median trawl CPUE and Jolly abundance index (± 2 standard errors) for Niantic River winter flounder from 1976 through 1986. Jolly index and CPUE adjusted for fish 15 cm and larger in 1983-86 and shown by * and Δ, respectively.

the catchability of marked and unmarked fish was not equal, sampling errors of unknown magnitude resulted. Jolly abundance estimates were also compared with those obtained using the Fisher-Ford and Manly-Parr models in 1980 and as results were generally similar, the simpler and more powerful Jolly model remained in use (NUSCo 1981a).

The latest review (NUSCo 1986a) rigorously examined the model itself, assumptions, reliability of results, and its particular application to the winter flounder population. The Jolly model has been criticized because its flexibility is provided by many unknown parameters which must be estimated. These estimates may be imprecise unless sampling intensity is relatively high (Cormack 1979; Buckland 1980; Nichols et al. 1981; Hightower and Gilbert 1984). The review concluded that sampling intensity and recapture rates

may be imprecise unless sampling intensity is relatively high (Cormack 1979; Buckland 1980; Nichols et al. 1981; Hightower and Gilbert 1984). The review concluded that sampling intensity and recapture rates of Niantic River winter flounder were rather low and may have resulted in errors of the estimates. Furthermore, estimates of recruitment (Jolly's B) were undoubtedly unreliable because they accumulated errors from other parameters in the model (also see Arnason and Mills 1981 and Hightower and Gilbert 1984). Adding recruitment estimates to an initial estimate of abundance (Jolly's N), as done in previous years to obtain absolute abundance estimates, was shown to be highly inaccurate. This led to the formulation of the composite index of abundance based on the Jolly model, which used the average of the central estimates of N and a pooled variance estimate (NUSCo 1986a).

Using results from a simulation study by Hightower and Gilbert (1984) and based on sampling intensities of 3 to 5.4%, population sizes of 20 to 40 thousand, and survival of about 0.90 ("deaths" here are mostly due to emigration from the Niantic River), it was assumed with 95% confidence that errors in estimating abundance using the Jolly model ranged from 25 to 50%. These levels of accuracy are sufficient for management purposes, but are not within the 10% level of error that Robson and Regier (1964) recommended for research investigations. Hightower and Gilbert (1984) demonstrated that sampling intensities had to be doubled to increase the accuracy from 25 to 50%. Substantially higher and most likely infeasible levels of effort would be required to achieve sampling intensities necessary for the accuracy level of 90% for current population sizes in the Niantic River.

The CPUE was also examined for bias and accuracy (NUSCo 1983b, 1986a). The standardization of fishing effort is difficult and although the catchability of a species is presumed constant, the fishing gear can vary in efficiency and can be affected by subtle changes in its use, rigging, or deployment from different vessels (Gulland 1983). Nevertheless, repeated surveys using relatively consistent methods can provide an index of abundance free of difficulties caused by possible changes in catchability. Among the factors examined that could have influenced Niantic River winter flounder CPUE were changes in boats, trawling methodology, and variable annual conditions in the Niantic River.

With some exceptions, the vessels used during sampling changed annually from 1976 through 1981 (Appendix XXIII). Several comparisons between boats showed significant differences in catches among years (NUSCo 1983b), indicating possible differences in fishing power. Since 1981, two vessels identical except for rigging have been used. *Northeast I*, rigged exclusively for trawling, consistently caught more

winter flounder in comparable tows than *Northeast II*, designed for lobster pot hauling and modified for trawling.

Another factor influencing CPUE was the distribution of effort among stations in the river (Appendix XXIV). Effort was most extensive in the lower river channel and adjacent shallows in 1976-77 and occurred mainly in the channel during 1978-80. Through 1980, it was believed that heavy concentrations of algae and detritus found in the upper river greatly reduced the number of winter flounder found there. However, in 1981 many winter flounder were found in the upper river and sampling effort has predominantly shifted there during the past 6 yr. Detrital loads probably also have altered the efficiency of the trawl and may have affected CPUE. Quantitative comparisons indicated significantly longer tow times in the upper river in 1984 than in 1983 or 1985-86. This was most likely due to the net filling with material as it was towed, thereby increasing drag. Decreases in average tow time have been due to lesser amounts of detritus as well as the purposeful avoidance of the worst towing areas in station 51. The practice of concentrating effort where most winter flounder were present prior to 1983 also may have had some consequence; increased medians may have resulted from limiting the number of tows with fewer fish.

Although each measure of abundance has inherent errors, the reasons for the disparities between the Jolly index and CPUE are unknown. The standardization of tows to a uniform distance by station in 1983 and subsequent years lessened the variability in CPUE as seen by the smaller confidence interval about each median and greater correspondence between the median and mean. The Jolly indices for 1984-86 were less precise with relatively large confidence intervals, which could have accounted for some of the differences observed in the trends. Because of the small number of fish marked in 1986, the Jolly estimate for that year may have been inaccurate. Although there were indications that the population probably declined from 1985, the decrease would have been proportionately less if the CPUE was more reliable than the Jolly composite index.

Some differences between CPUE and the Jolly composite index could be related to changes in winter flounder distribution over time. Possibly, most winter flounder were present only in the lower river during the 1970s and despite low to moderate population levels, high CPUE were obtained because of their concentration in the relatively smaller portion of the river that was sampled. Then, in 1980, for unknown reasons, winter flounder began to use the upper river as well. Although more abundant, they would have been less concentrated throughout the river and lower CPUE were obtained. Alternatively, winter flounder

may have always been present throughout the river and the Jolly indices from 1977 through 1980 were inaccurately low to an unknown degree because only a portion of the population was sampled. This was previously suspected for the 1980 survey, when a large percentage of relatively small and stationary winter flounder were found and no samples were taken in the upper river unlike following years (NUSCo 1983b, 1985). Therefore, CPUE may have more realistically measured abundance from 1976 through 1980. Examination of data from 1981 through 1986 showed that median CPUE values for station 1 and 2 in the lower river were very similar to those from the upper river (within 4 fish per tow), except for 1983 (13.2 less in the lower river), so CPUE appears to be relatively consistent among areas. As surveys and methods are presently more standardized, further comparisons of CPUE and the Jolly index may be made in forthcoming years and the relative accuracies of the two measures of abundance can be reassessed.

Harmonic regression models

Another measure of winter flounder abundance throughout the Millstone area was the development of time-based harmonic regression models. Log-transformed data from six stations of the trawl monitoring program (see Fish Ecology section) were used to describe the fluctuations in abundance of winter flounder. Models having data from October 1976 through September 1984 forecasted catches from October 1984 through September 1985 and the actual catches were then compared to those predicted (NUSCo 1986a). Similar models were reported in NUSCo (1984, 1985) for 1982-83 and 1983-84 data. Results showed that models for stations other than NR (Niantic River) were not satisfactory. Terms corresponding to a sine-cosine function describing an 8-yr period were significant for most models in 1985, as were terms for 7 or 6 yr during 1984 and 1983. Since these terms represented the entire time-series of data at the time of model development, this indicated either insufficient data or a lack of a repetitive pattern of abundance. Terms of less than 1 yr were also found for most models and were probably related to annual cycles of abundance due to local movements and recruitment of juveniles into the trawl catch. Except for NR (0.71), R^2 values for each model remained low (0.38-0.53). However, this is a typical result for a species taken year-round in samples. Forecast errors remained high in 1985 (65-279%; 128% for NR), although, in most cases, improved each year since 1983. This was an indication that the models were perhaps providing better predictability.

Many factors have influenced trawl monitoring program catches, as they have in the Niantic River abundance surveys discussed previously. Despite its relatively high abundance in the catch (43% of the

10-yr total, ranked first), typical high variability along with relatively low effort makes analyses of these data problematical. In addition, the mixture of stocks present at most stations during many months of the year (see Stock Identification below) makes data from the trawl monitoring program difficult to interpret and of limited use in assessing the impact of MNPS operations.

Regional trends in abundance

Data from other sources were examined for comparisons in abundance as historically the abundance of winter flounder has been known to fluctuate, showing various periods of increases and decreases (Perlmutter 1947; Howe 1975; Ketschke 1977; Jeffries and Terceiro 1985). This feature of winter flounder population dynamics, demonstrated above for the Niantic River population, also was found to have occurred in other areas of Southern New England. Jeffries and Terceiro (1985) reported that winter flounder abundance at a station in Narragansett Bay decreased 86% from 1968 to 1976, but increased rapidly to reach another peak in 1979. This was followed by a steady decline through 1982. Flounder abundance at another station in Rhode Island Sound showed similar fluctuations, indicating that the changes in abundance were not due to a shift in population from inside to outside of the bay. However, these finding were based on only one tow per week at each station.

Commercial landings throughout Southern New England have decreased steadily from 11,100 mt in 1981 to 7,000 mt in 1985, although they have remained higher than any year prior to 1980 (NMFS 1986). Commercial vessel CPUE fell to a historical low in 1985 from a peak in 1981. The National Marine Fisheries Service (NMFS) offshore trawl surveys found that survey vessel CPUE decreased rapidly from a maximum in 1981 (3.6 kg/tow) to levels below the long-term average in 1984 (0.8) and 1985 (1.0). Based on these declines, NMFS considers the winter flounder to be fully exploited and that current catch levels will probably not be sustained (NMFS 1986).

The Massachusetts coastwide fishery assessment fall survey found an 84% reduction in winter flounder biomass from 1983 to 1984 (Howe et al. 1985). Commercial landings also have declined 52% in Massachusetts since 1981. Some evidence suggests that overfishing of winter flounder has occurred in Massachusetts (MDMF 1985). Besides decreases in landings, percentages of market-sized "small" and "pee-wee" fish have increased; relatively high landings were maintained by landing more and more smaller fish. The

percentage of fishermen directing their effort towards winter flounder and shift of larger vessels inshore has exacerbated the problem and serious concern was expressed about the fishery.

The annual geometric mean trawl catch of winter flounder by the CT DEP in eastern Long Island Sound declined fivefold from 1984 through 1986 (P. Howell, CT DEP, pers. comm.), a decrease even greater than seen in the Niantic River. Conversely, annual landings from Connecticut-licensed commercial trawlers increased threefold from an average of 87,657 kg during 1979-82 to 271,160 kg during 1983-86 (CT DEP, unpublished data). It should be noted that some of the increase in landings in recent years was due to more accurate reporting by fishermen (E. Smith, CT DEP, pers. comm.). From 13 to 43% of these fish were caught in eastern Long Island Sound, with increasing proportions (27-43%) for recent years. Percent landings from outside Long Island Sound decreased during the same period from more than half to about one-third of the total. At the same time, overall catch-per-trawl-hour decreased from an average of 41.9 kg during 1979-83 to 30.0 kg in 1984-86.

Life history studies

Reproduction

Sex ratio

The sex ratio of winter flounder larger than 20 cm during the spawning period in the Niantic River varied from 0.92 to 2.03 females for each male (Table 9). The average from 1977 through 1986 was 1.44; the latest survey was the only one in which more males than females were taken. Sex ratios of 1.50 to 2.33 in favor of females were also reported by Saila (1962a, 1962b) and Howe and Coates (1975) for other populations in southern New England.

Size at maturity

Female winter flounder can become sexually mature when they are age 3 or when about 20 cm in length (Dunn and Tyler 1969; Dunn 1970; Kennedy and Steele 1971; Beacham 1982). More northerly populations mature at smaller sizes and older ages than in Southern New England. Results of a probit

Table 9. Female to male sex ratios of winter flounder taken during the spawning period in the Niantic River from 1977 through 1986.

	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	Mean	C.V.
All fish captured	1.03	2.23	1.37	2.66	1.42	1.16	1.52	1.07	1.37	0.92	1.48	38%
Measured fish > 20 cm	1.26	1.95	1.21	2.03	1.61	1.50	1.52	1.07	1.37	0.92	1.44	25%

analysis showed that the length of 50% sexual maturation of Niantic River females during 1981-86 was 26.8 cm with a 95% confidence interval of 26.3 to 27.3 cm. On an annual basis, values ranged from 25.1 cm in 1983 to 29.4 cm in 1981. Most of these fish were age 3 or 4. Males can mature at even smaller sizes (down to 10-12 cm) and earlier ages (2) than females.

Spawning

Winter flounder spawning was followed by noting the weekly change in the percentage of gravid females larger than 25 cm in the Niantic River. Generally, most spawning was completed by early April (Fig. 5). Ice in the river prevented starting population surveys earlier in January or February, so for most years approximately two-thirds of the females examined during late February had spawned before sampling began. In most cases, spawning appeared to be correlated with water temperature. In relatively cold years (1977-78, 1982), fewer females spawned during the earlier portion of the survey, whereas in warmer years (1981, 1983-86) more were spent. No conclusions can be made about 1979 or 1980 (both cooler than average) because of the late starts to these surveys and because relatively fewer females were examined in comparison to other years.

Fecundity and egg production

The length-fecundity relationship for winter flounder taken in the vicinity of MNPS was determined in two independent studies during 1977. In one, all fish were collected in the Niantic River and in the

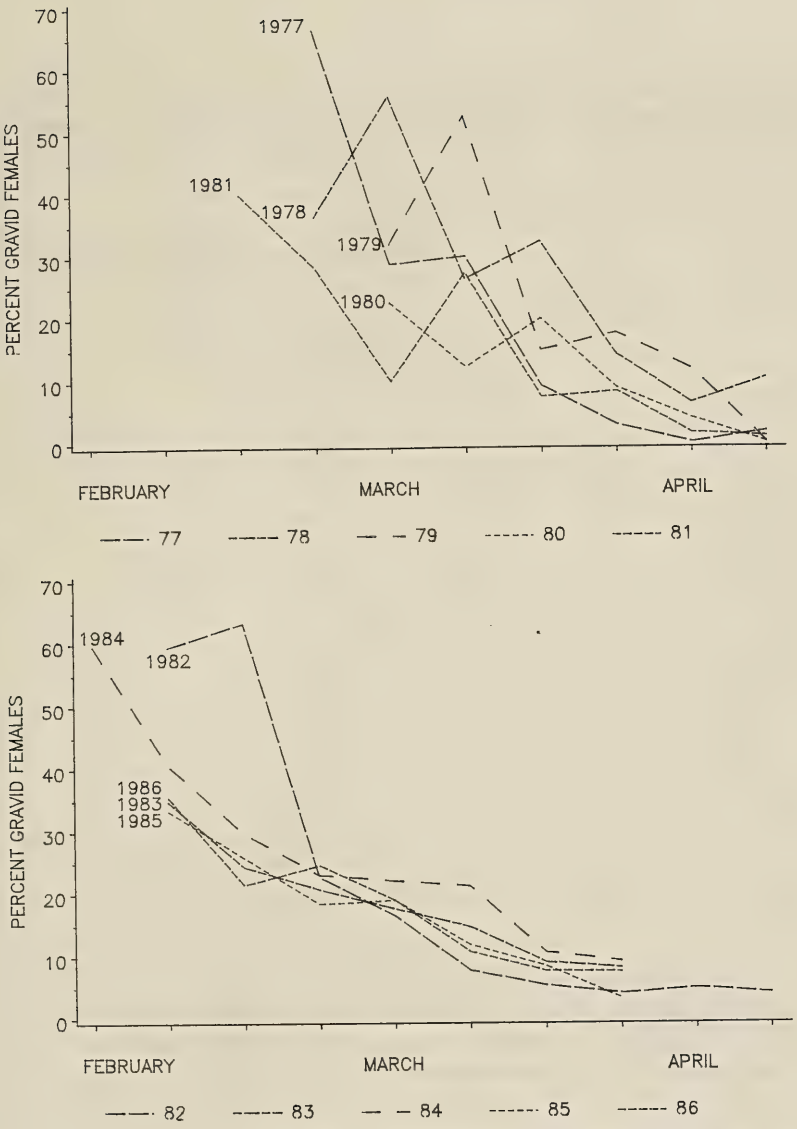


Figure 5. Percentage of adult female winter flounder in spawning condition by week in the Niantic River from 1977 through 1986.

second they were caught by trawl in Niantic Bay or were impinged at MNPS. Laboratory methods and procedures for determining fecundity differed in some respects between the studies. Although findings

Table 10. Length-fecundity and length-weight relationships for winter flounder.

Sample location	Type regression	n	Slope (b)	Intercept (a)	r^2	Value for 31-cm fish
			Length-fecundity[b]			
Niantic River	Functional	48	4.5060	-1.0842	0.81	431,000
Niantic Bay	Functional	65	3.9708	-0.2035	0.84	523,000
Rhode Island (Saila 1962a)	Functional[a]	45	4.6043	-1.2542	0.63	410,000
Newfoundland (Kennedy and Steele 1971)	Least squares	23	4.0631	-0.4517	?	405,000
			Length-weight[c]			
Niantic River	Functional	270	3.2261	-2.2261	0.99	385 g
Niantic Bay	Functional	491	3.3391	-2.2411	0.94	548 g
Mass.-R.I (Lux 1969)	Least squares	2,118	3.138	-5.239	?	379 g
Newfoundland (Kennedy and Steele 1971)	Least squares Females: Males:	 117 107	 3.1441 2.9833	 -2.0702 -1.9041	 ? ?	 416 g 351 g

[a] Recalculated from Saila's (1962a) published data.

[b] Regression equation in the form: $\log_{10}$ fecundity = a + b($\log_{10}$ length in cm).

[c] Regression equation in the form: $\log_{10}$ weight (g) = a + b($\log_{10}$ length in cm) (except Lux 1969, where length is in mm).

based on these data were reported previously (NUSCo 1978a, 1983b), they were re-examined for this report and new length-fecundity relationships were calculated using a functional regression (Table 10). Using the 1977-86 modal length of 31 cm for Niantic River females for comparison, the fecundity estimate obtained from the first study was most similar to estimates made using relationships obtained from Saila (1962a) and Kennedy and Steele (1971). The Niantic Bay data gave comparatively high estimates, but no reason can be given for the discrepancy between the two studies.

Table 11. Annual indices of female spawners and egg production for Niantic River winter flounder from 1977 through 1986.

Year	Number of spawning females[a] ($x10^3$)	% of population comprised by mature females[b]	Mean fecundity ($x10^5$)	Total egg production index ($x10^9$)
1977	5.6	31%	4.8	2.69
1978	5.5	38%	5.1	2.81
1979	4.8	26%	5.3	2.55
1980	3.7	20%	4.7	1.70
1981	10.7	37%	5.3	5.66
1982	18.9	38%	5.7	10.79
1983	15.2	51%	5.6	8.51
1984	12.6	43%	5.7	7.20
1985	10.1	47%	5.9	5.92
1986	3.5	43%	6.5	2.31

[a] From composite index of abundance and percentage of mature females, assuming that all females 26 cm and larger were mature.

[b] For winter flounder larger than 15 cm during 1976-82 and 20 cm thereafter.

The proportion of females in the population larger than 26 cm was combined with the Jolly index of abundance to obtain a relative number of female spawners from year to year; 1976 was excluded because no reproductive data were available then. Spawning females comprised 20 to 51% of the population (Table 11). Percentages for 1977-82 were lower because they were based on all fish larger than 15 cm and thus included more immature fish in the abundance estimates. The value for 1980 (20%) was particularly low and is another indication that a segment of the spawning population was missed then, as comparatively more smaller, immature fish were caught.

The mean fecundity was calculated using the relationship described previously with annual length-frequency data. Values have been relatively consistent with somewhat greater means found since 1982, when the start of the surveys was advanced into February. During the past several years, most females larger than 40 cm were usually found in the Niantic River early in the season and many evidently left the estuary in March. Surveys during earlier years started after February and probably missed many of these large winter flounder, resulting in a lower mean fecundity. The mean length of all females 20 cm and larger in 1980 was only 29.7 cm, in comparison to 31.4 to 32.1 cm for 1979 and 1981-85. This was additional evidence that larger females were missed during the 1980 survey. In contrast, the 1986 mean of 33.4 cm was particularly large, indicating that along with decreasing abundance, the female population was comprised of relatively larger and older specimens.

Egg production indices were determined using Jolly abundance indices with the length, maturity, and fecundity data. Since the indices reflect both the annual mean fecundity and abundance, the value for 1980 was probably underestimated. The egg production index peaked in 1982 and has declined about 80% since then. Tyler and Dunn (1976) reported that the relationship between length and egg production of New Brunswick winter flounder varied from year to year, depending upon variable nutrition and females were found to sacrifice egg production to maintain body weight. It is not known whether fecundity varies annually among Niantic River winter flounder.

Age and growth

Length-weight relationship

As with fecundity, the length-weight relationships for winter flounder taken in Niantic River and Bay were recalculated using 1977 data with a functional regression. Once again, the relationship determined for Niantic River fish appeared to be more consistent with other published regressions (Table 10). The Niantic Bay relationship gave heavier weights per unit of length; the reason for the difference is unknown.

Age and length

In 1977 and 1978, winter flounder were aged by examination of both scales and otoliths; age determined by both structures agreed well, especially for younger fish (NUSCo 1979). As the use of otoliths required sacrificing the fish, only scales were used in subsequent years. The overall mean lengths-at-age for 1977-1983 combined data showed that winter flounder grew most rapidly during the first several years of life with considerably reduced annual growth after age 4 (Table 12). Females had greater annual increases in length than males, especially at ages 3 through 5. The mean lengths of females were significantly greater than males for fish age 3 and older; this was also noted for many other winter flounder populations (Berry et al. 1965; Poole 1966; Lux 1973; Howe and Coates 1975; Danila 1978; Beacham 1982).

Table 12. Mean lengths in mm by age of Niantic River winter flounder from 1977 through 1983.

Age	Sex	Number	Mean	SD	CV	Lower quartile	Upper quartile	Tenth percentile	Ninetieth percentile
1	?	1,259	82	23	28%	65	97	55	118
2	?	1,413	176	27	15%	156	195	142	214
3	?	139	212	15	7%	201	223	192	235
3	F	460	266	27	10%	245	284	234	304
3	M	220	256	28	11%	237	278	220	292
4	F	246	307	29	10%	289	327	267	344
4	M	159	291	26	9%	275	308	260	320
5	F	174	342	24	7%	325	360	314	370
5	M	150	312	26	8%	295	330	280	358
6	F	112	363	22	6%	345	379	333	391
6	M	90	336	25	8%	316	351	303	370
7	F	78	390	23	6%	374	405	362	419
7	M	57	360	25	7%	337	378	331	392
8	F	51	409	27	7%	391	431	378	442
8	M	18	368	31	8%	345	386	327	426
9	F	13	428	26	6%	408	447	389	467
9	M	4	400	29	7%	--	--	369	437
10	F	4	431	17	4%	--	--	422	456
10	M	1	411	--	--	--	--	--	--
12	F	1	443	--	--	--	--	--	--

The coefficients of variation for length by age were relatively consistent and ranged from 6 to 11% for age 3 through 8 females and males. Using the lower and upper quartiles as dividing points, age-length groupings were relatively distinct, especially for younger specimens. However, older age groups overlapped considerably in length. The oldest specimens aged were an age 12 female and an age 10 male. Maximum lengths recorded in the Niantic River included a 490-mm female and a 483-mm male, both in 1986. However, in May 1981, a 508-mm female determined to be age 8 was impinged on the MNPS screens. Based on its length and age, this specimen fit the growth curve for the Georges Bank stock of winter flounder described by Lux (1973).

For the purposes of calculating growth, 214 females and 188 males ranging from 44 to 465 mm were aged by scale examination in 1983 and measurements were made to each annulus (NUSCo 1984). A non-linear length-scale relationship was used for the back-calculation of length-at-age because it provided a better fit to the data. Except for ages 1 and 2, the mean calculated lengths-at-age of females were usually larger than observed lengths (Table 13). The trend was not as obvious for males (Table 14). Calculated growth estimates were probably less reliable for older specimens, particularly males because of small sample size. A reverse Lee's phenomenon (Tesch 1968; Ricker 1975) was observed in which the calculated lengths of fish increased as the age of the fish increased. This may have been the result of size-selective mortality that was greater on the smaller fish of an age group (Tesch 1968) or due to a bias in sample selection if only faster-growing larger specimens were used in aging and scales from slower-growing fish were rejected as unreadable. This may also have resulted in the apparent anomalous increase in growth at age 9 in females and 8 in males.

Growth of the Niantic River stock was compared in Figure 6 to that of other populations in nearby areas, including Charlestown Pond, RI (Berry et al. 1965), Peconic Bay, NY (Poole 1966), and south of Cape Cod, MA (Howe and Coates 1975). The Niantic River fish grew less through age 2 than these and other populations (Poole 1966; Kurtz 1975; Danila 1978), with the exception of the nearby Mystic River (Pearcy 1962). However, growth of Niantic River fish equaled or exceeded that of other stocks age 3 and older. Although the winter flounder is an omnivorous feeder (Pearcy 1962; Richards 1963), conditions in the Niantic River may not be as favorable for the growth of immature fish as other areas.

Table 13. Average back-calculated lengths (mm) at age for female winter flounder taken in the Niantic River.

Age class	Number	Mean length at capture ± 95% CI	Mean calculated length (± 95% CI) Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9
I	30	107±14	80±8								
II	26	195±13	67±9	178±16							
III	43	277±11	78±7	182±12	285±12						
IV	25	315±13	87±10	185±14	278±12	318±14					
V	23	349±10	88±9	182±16	276±18	321±18	343±18				
VI	13	366±19	75±15	163±23	269±21	309±16	331±15	345±14			
VII	27	395±10	90±7	205±13	269±14	332±16	354±17	369±18	380±18		
VIII	19	418±9	86±11	202±24	296±25	345±21	370±21	383±21	393±21	402±22	
IX	8	442±10	86±10	186±26	292±33	361±26	389±25	406±27	419±25	428±27	435±26
Mean calculated length			81±4	186±6	285±6	329±8	355±8	373±10	391±12	410±16	435±26
Average growth increment			81	105	99	44	26	18	18	19	25

Table 14. Average back-calculated lengths (mm) at age for male winter flounder taken in the Niantic River.

Age class	Number	Mean length at capture ± 95% CI	Mean calculated length (± 95% CI) Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8
I	28	96±13	80±8							
II	18	177±16	70±13	166±19						
III	43	246±10	81±7	183±10	262±12					
IV	19	285±15	75±10	171±19	248±20	281±21				
V	26	327±12	92±12	181±18	259±18	301±16	322±16			
VI	20	348±12	84±9	185±17	257±18	302±17	322±17	334±18		
VII	27	370±10	81±9	189±20	260±21	300±21	320±20	337±19	349±19	
VIII	7	378±28	91±17	209±28	305±46	351±45	369±42	380±41	389±40	397±39
Mean calculated length			82±4	182±6	261±8	302±10	325±10	341±12	357±18	397±39
Average growth increment			82	100	79	41	23	16	16	40

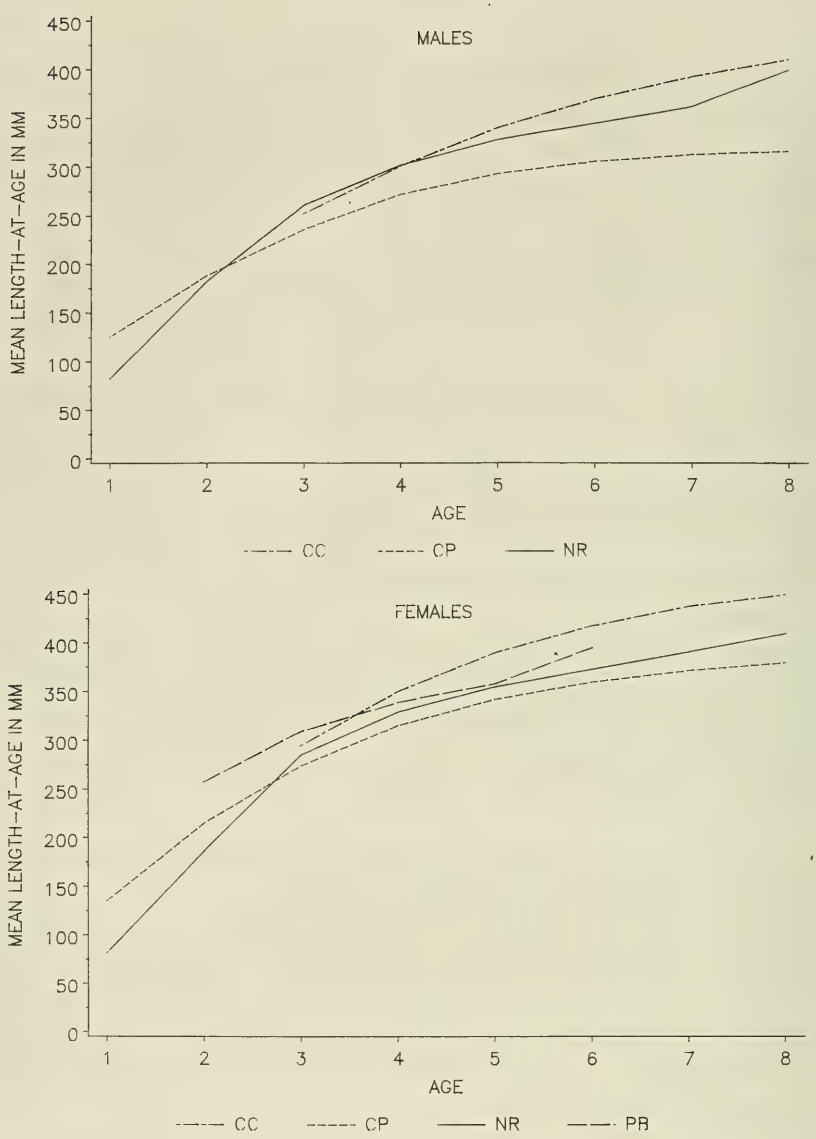


Figure 6. Calculated mean lengths of winter flounder by age from various locations in the northeastern United States (CC = south of Cape Cod, CP = Charlestown Pond, NR = Niantic River, PB = Peconic Bay).

von Bertalanffy growth model

Calculated-lengths-at-age from 1983 data were used to estimate the von Bertalanffy growth parameters (Table 15). The growth models for females and males had good fits to the data and represented theoretical growth of the population. Use of a nonlinear procedure for parameter estimation should have resulted in the least biased and variable estimates in comparison with traditional linear methods used in most older studies (Vaughan and Kanciruk 1982). The ω parameter was used for comparisons of growth as suggested by Gallucci and Quinn (1979). They suggested constructing a rectangle formed by two standard errors on each side of point estimates of L_∞ and K. This was done for both sexes and because the rectangles did not overlap, significant differences in growth were indicated between females and males. It was not possible to make comparisons with other winter flounder populations because of the lack of published estimates of variability. However, the ω parameter of female Niantic River winter flounder was similar to those of other stocks or geographical groups examined with the exception of Georges Bank, which has a racially distinct population with much greater growth (Lux et al. 1970; Lux 1973; Howe and Coates 1975). The value for males most closely corresponded to the Charlestown Pond stock (Berry et al. 1965); greater asymptotic maximum length was achieved by winter flounder stocks in other areas to the east.

The estimates of L_∞ were actually less than the lengths of some specimens examined in the Niantic River. However, this should not be considered unusual as L_∞ represents the maximum length that an average fish would achieve if it grew indefinitely (Fabens 1965; Ricker 1975). This could have been a result of the particular sample used and inclusion of additional larger and older specimens could have increased asymptotic length estimates. Nevertheless, since 1977 lengths of only 1.7% of 14,374 females and 1.0% of 10,706 males larger than 20 cm exceeded the calculated values.

The 1977-83 age-length data were also fit to the von Bertalanffy model (Table 15). Because sex of age 1 and 2 and smaller age 3 fish was not ascertained in the field, these specimens were used separately with both older females and males in fitting the model. Although growth of females and males was found to be similar in early life, it is not known how the calculation of model parameters was affected by using these data with each sex. Using the 1977-83 data, significantly larger estimates of L_∞ (453 mm for females; 397 for males) and smaller values of K (0.30; 0.34) were obtained in comparison with results using the 1983 calculated length-at-age data. Another error potentially affected results as the observed lengths were taken over a period of several months each year and individuals could have been actually older or younger

Table 15. The von Bertalanffy growth parameters for Niantic River winter flounder and comparisons with other stocks.

Area	No. of fish examined	K value	Asymptotic 95% CI	$L\infty$ (mm)	Asymptotic 95% CI	ω^a	t_0 (year)	Asymptotic 95% CI	R^2
Females									
Niantic River[b]	214	0.42	0.39-0.45	423	412-433	177.7	+0.51	0.46-0.56	0.90
Niantic River[c]	3789	0.30	0.29-0.31	453	446-461	135.9	+0.35	0.33-0.37	0.93
Niantic River[d]	129	0.35	0.27-0.44	423	398-448	148.1	--	--	0.90
Charlestown Pond[e]	104	0.41	--	396	--	162.4	--	--	--
South of Cape Cod[f]	839	0.34	--	487	--	165.6	--	--	--
North of Cape Cod[f]	114	0.37	--	455	--	168.4	--	--	--
Georges Bank[f]	126	0.44	--	622	--	273.7	--	--	--
Georges Bank[g]	163	0.31	--	630	--	195.3	-0.05	--	--
Males									
Niantic River[b]	188	0.44	0.39-0.48	381	367-395	165.7	+0.46	0.39-0.53	0.85
Niantic River[c]	3286	0.34	0.33-0.35	397	389-405	135.0	+0.34	0.31-0.36	0.91
Niantic River[d]	81	0.31	0.23-0.39	375	357-392	116.3	--	--	0.87
Charlestown Pond[e]	49	0.54	--	323	--	174.4	--	--	--
South of Cape Cod[f]	298	0.25	--	477	--	119.3	--	--	--
Georges Bank[f]	113	0.37	--	534	--	197.6	--	--	--
Georges Bank[g]	184	0.37	--	550	--	203.5	+0.05	--	--

[a] $\omega = K \times L\infty$ (Gallucci and Quinn 1979)

[b] Fit to 1983 calculated lengths-at-age.

[c] Fit to 1977-83 age-length data.

[d] From method of Fabens (1965) using data from recaptured disc-tagged fish.

[e] Berry et al. (1965); parameter K calculated from their k ($= e^{-k}$)

[f] Howe and Coates (1975); parameter K calculated from their K

[g] Lux (1973)

than the stated age. Conversely, error may have been introduced using the calculated lengths-at-age because of variable timing of annulus formation. Also, if mostly faster-growing older fish were selected for measurements of annuli on scales, as suggested previously, a larger K value may have resulted because these specimens had a faster rate of growth.

An independent assessment of growth was obtained from winter flounder marked with Petersen disc tags, released in the study area, and later recaptured after various periods at large. The fish used in this analysis included females from 201 to 406 mm and males from 205 to 398 mm at time of tagging. The recaptured fish were caught after 90 to 1,065 d at large and growth ranged from 1 to 149 mm in females and 1 to 85 mm in males. The data were fit to a two-parameter (K and L_∞) von Bertalanffy model described by Fabens (1965) for fish of unknown age, but whose increase in length is known for varying time periods. The estimates of L_∞ (423 mm for females; 375 for males) were similar to those described by the 1983 calculated length-at-age data, but the K values (0.35; 0.31) were closer to the 1977-83 aging data model. However, less confidence can be placed on the parameters determined from the tagged fish due to variability in the data and smaller sample size. Growth of individuals also varied because of the particular season of release or recapture and effects of tagging may have influenced these results.

Mortality and survival

Mortality and survival were calculated using age and length data from 1978-79 and 1981-83. Data from 1977 were not used because relatively few fish were aged and from 1980 because of previously mentioned inconsistencies of data from that survey. Both methods of estimation used were time-specific (Ricker 1975). Apparent large variability in estimated abundance of individuals of specific year-classes, most likely due to changes in survey methodology, sample selection, and sampling error, made cohort-specific methods of estimating survival less reliable or simply not possible (NUSCo 1984). The time-specific method of Robson and Chapman (1961) has an advantage in that the age determinations of older fish do not have to be known with certainty, although the representativeness of the youngest age used is very important (Ricker 1975). As noted previously, relative abundance of small winter flounder (ages 1 and 2) in the population age structure were not accurately measured. Therefore, only ages 3 and older were used in the calculations.

The geometric mean annual survival rate (S) for the five estimates was 0.572, corresponding to an instantaneous mortality rate (Z) of 0.558 (Table 16). The value is larger than the survival estimate of 0.526 determined by Howe and Coates (1975) for winter flounder south of Cape Cod and even greater than that for fish from Great South Bay, N.Y. (0.27-0.28; Poole 1969) and Rhode Island (0.35-0.49; Berry et al. 1965). The estimate may be biased because of violations in required assumptions. Ricker (1975) noted that Robson and Chapman's formula assumes that survival rate is constant at all ages, that all year-classes are recruited at the same abundance, and that all ages are equally vulnerable to the sampling gear. When these conditions are not met, estimates of S are biased and confidence intervals are too narrow. Ricker also noted that differences in year-class strength are usually found for most stocks and that the best estimate of S will be made using a catch curve with equal weighting.

Accordingly, catch curves were constructed for each year of data. Examination of the plotted log frequencies of age showed that the curves were nonlinear. Furthermore, some bumps in the curves increased with year, indicating probable non-uniform recruitment with stronger year-classes influencing catches more so than average or weak ones. Tendencies for age 3 and age 7 and older frequencies to be lower than the predicted regression line also suggested less than complete recruitment at age 3 and increasing mortality in older age groups. Even so, for all years examined, the mean age of specimens age 3 and older ranged only from 4.1 to 4.5. Annual estimates of Z ranged from 0.515 to 0.708, with a geometric mean of 0.611.

Table 16. Survival (S) and instantaneous mortality rate (Z) of age 3 and older Niantic River winter flounder determined using the method of Robson and Chapman (1961).

Year	S	95% CI	A[a]	Z
1978	0.549	0.528-0.569	0.451	0.600
1979	0.564	0.539-0.589	0.436	0.573
1981	0.574	0.558-0.589	0.426	0.555
1982	0.567	0.552-0.581	0.433	0.568
1983	0.608	0.600-0.616	0.392	0.498
Geometric mean	0.572	---	0.428	0.558

[a] Annual mortality rate = 1-S

Following Ricker's suggestion to combine samples from successive years, catch curves were constructed from combined data for 1978-79 and 1981-83 (Fig. 7). Catches within each group were adjusted to give equal weight to each year. Although years were combined, irregularities were still seen in the curves. The geometric mean of Z for the two yearly groups was 0.721, corresponding to a survival rate of 0.486 and an annual total mortality rate (A) of 0.514. These rates are probably less biased than the estimates made using the method of Robson and Chapman and represent the best estimate of mortality for Niantic River winter flounder.

Food habits

Major taxonomic groups of organisms eaten by winter flounder in the vicinity of MNPS were examined from June 1973 through November 1974 (NUSCo 1975). Important food items consumed included crustaceans, annelid and polychaete worms, and mollusks (Table 17). Algae was also frequently found in stomachs at several stations, supporting the contention of Wells et al. (1973) that filamentous algae was frequently eaten by winter flounder. This separates the winter flounder from most other northeastern Atlantic marine fishes, which are strictly carnivorous. Average stomach fullness ranged from about one-third full in the Niantic River to half-full at Seaside Point and Twotree Island Channel. Food items varied by location and seemed to reflect bottom type and different benthic communities. For example, bivalves were particularly important in the muddier upper Niantic River, whereas worms and crustaceans were more important in the sandier areas of the lower river and in Niantic Bay. The variable diet of winter flounder in the area was not unexpected as it has been reported to be an omnivorous, opportunistic feeder (Pearcy 1962; Richards 1963; Mulkana 1966; Frame 1972; Kurtz 1975; Festa 1977; Scarlett 1986). Richards (1963) reported that winter flounder fed on more prey taxa than any other demersal fish in Long Island Sound. Several studies have hypothesized that some movements of winter flounder are feeding migrations not associated with water temperature preferences or for spawning (Kennedy and Steele 1971; Van Guelpen and Davis 1979). However, not enough data were available for Niantic River winter flounder to examine this possibility.

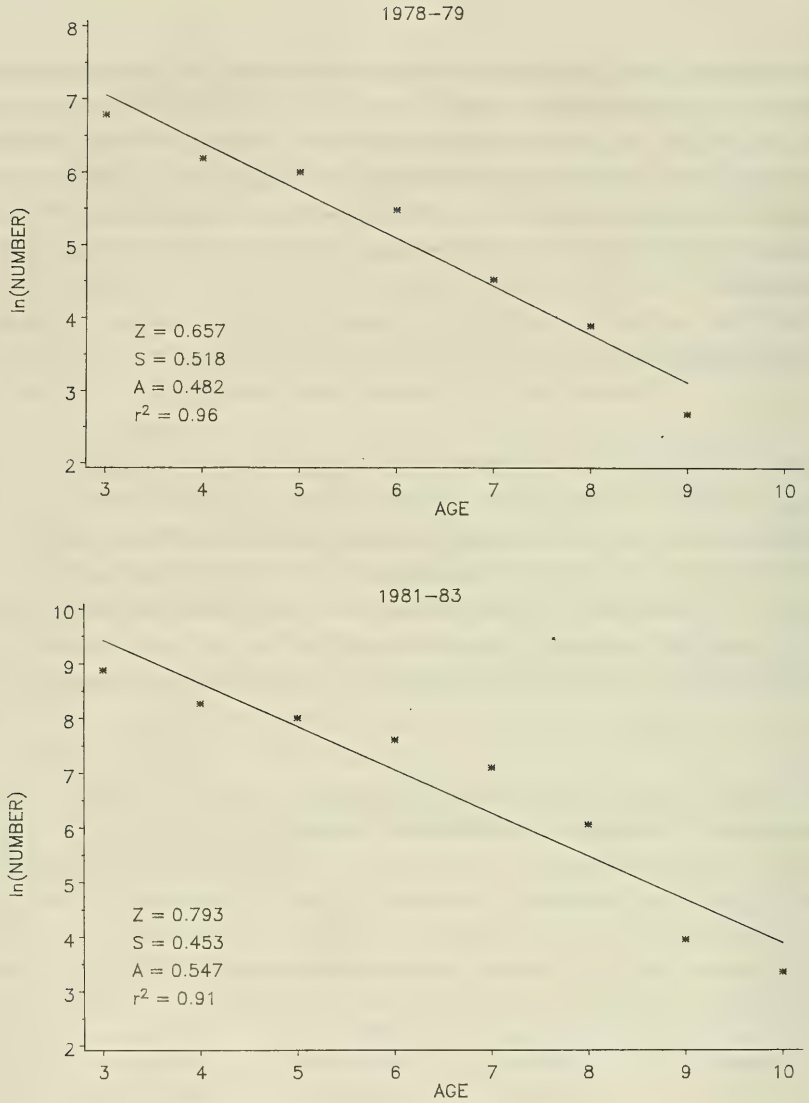


Figure 7. Mortality determined by catch curve for Niantic River adult winter flounder from 1978-79 and 1981-83.

Movements and exploitation

Some tag return data from 1973 and 1974 studies were reported in NUSCo (1975). Anchor tags were used in an attempt to estimate abundance and recaptures were made throughout the year during sampling activities and by fishermen. Of 566 recaptures with location known, 422 (75%) were from local waters in Niantic Bay and River. Most (81%) of the remainder were caught from 5 to 15 mi east of MNPS, 8 (10%) from within 5 mi to the west, 3 (4%) from 30 to 50 mi west, and 4 (5%) from offshore waters. A large majority (80%) were caught during March through May of 1973 and 1974. Few tags were received from commercial fishermen which may be related to relatively poor retention of this type of tag in flatfish (J. Castleman, NUSCo, pers. comm.), or to less cooperation in making returns.

Table 17. Average estimated stomach fullness by major prey of winter flounder taken in the Millstone Point area from June 1973 through November 1974.

	Area						
	Upper Niantic River	Niantic Bay	Jordan Cove	Seaside Point	Twotree Island Channel	Black Point	Bartlett Reef
Station bottom type	mud	mud and sand	sand	coarse sand and mud	sand	sand	sand and rock
Number of fish examined	39	48	51	32	49	49	38
			Prey organisms				
Algae	3.4	1.7	12.4	16.1	4.9	1.8	1.6
Sponges	--	0.1	--	--	0.2	1.2	2.4
Bryozoans	1.8	--	--	1.2	1.1	2.7	5.6
Mollusks	--	0.3	--	--	--	--	--
Univalves	2.6	--	1.6	0.4	0.2	--	--
Bivalves	14.9	4.2	1.8	2.5	15.1	9.9	7.4
Annelids	2.2	6.4	1.0	1.9	0.7	1.1	0.5
Polychaetes	4.7	6.4	14.8	14.2	5.6	10.8	8.4
Crustaceans	5.1	14.9	6.4	8.3	25.0	13.7	14.1
Crabs	0.5	2.5	0.7	4.2	0.5	1.5	--
Shrimp	0.7	--	0.2	--	--	--	0.2
Fish	--	--	--	--	--	0.3	--
Miscellaneous or unidentified	0.8	2.2	2.6	1.3	1.4	2.7	2.8
Average total points[a]	36.7	38.7	41.5	50.1	54.7	45.7	43.0

[a] 100 = full, 50 = half full, 0 = empty

A study designed specifically to determine movements and exploitation by the sport and commercial fisheries was undertaken from December 1980 through September 1983. Each of 4,978 specimens larger than 20 cm was tagged with a Petersen disc (Table 18). Most were released in the Niantic River (46%) and Bay (31%). About 61% were females, 29% were males, and 10% were fish whose sex was not determined, mostly because of their smaller size.

More than half (57%) of the 1,227 recaptures were made within 6 mo of release (Table 19). Larger percentages of fish were taken more quickly by NUSCo sampling than by the fisheries because sampling was concentrated in the Niantic River shortly after most were released. Also, tagged fish set free in special studies near the MNPS intakes tended to be impinged quickly, if at all. Recaptures dropped off rapidly after 1984 with only 20 returns for 1985 and 2 in 1986. None of the latter fish were tagged in 1980 or 1981; about 40% were released in 1982 and 60% in 1983.

The overall rate of return through 1986 was 25%, with 40% of the recaptures made by sport fishermen, 33% from NUSCo sampling activities, 24% from the commercial fishery, 2% from fish impinged on the MNPS traveling screens, and 1% from miscellaneous sources. This suggested that the sport fishery is a significant source of mortality for the Niantic River winter flounder, also noted by Sampson (1981) and Blake and Smith (1984). However, judging by the number of tags received from fish markets and processing plants, cooperation by commercial fishermen in returning tags was most likely less than for sport fishermen. The rates of return for each tagging area (ignoring NUSCo recaptures because of the disproportionate effort in the Niantic River) were similar and ranged from 11% for fish released at Bartlett Reef and the MNPS intakes to 16-17% for Niantic River and Bay and Twotree, and 26% for Jordan Cove.

The returns from the sport and commercial fisheries varied by location of release. The sport fishery took about three-quarters of the recaptured fish that were released in the Niantic River and Jordan Cove embayments. The return from the Niantic Bay and Twotree releases was nearly equal between the two fisheries, but about twice as many fish were caught by the commercial as the sport fishery from the deeper water Bartlett Reef station.

Sixty-five percent of the recaptures were female, 31% were male, and 4% were fish of undetermined sex. Similar (27%) proportions of all males and females released were caught again, but only 10% of the undetermined fish were recaptured. It was suggested in NUSCo (1985) that because the latter fish were

Table 18. Summary of winter flounder disc tagging and recapture data from December 1980 through September 1986.

		Niantic Bay	Niantic River	Twotree	MNPS Intakes	Barlett Reef	Jordan Cove	Misc.[a]	Total
Number tagged	1980	309	410	47	0	68	10	0	844
	1981	970	108	29	198	4	0	0	1,309
	1982	280	1,015	294	61	206	131	10	1,997
	1983	0	770	0	40	12	6	0	828
	Total	1,559	2,303	370	299	290	147	10	4,978
Number recaptured[b]	1980	74	89	15	0	8	0	0	186
	1981	179	22	3	27	0	0	0	231
	1982	55	347	56	11	22	46	2	539
	1983	0	264	0	4	2	1	0	271
	Total	308	722	74	42	32	47	2	1,227
Method recaptured:									
Sport fishing	1980	31	28	7	0	1	0	0	67
	1981	64	8	0	8	0	0	0	80
	1982	29	146	21	5	9	28	1	239
	1983	0	105	0	0	0	0	0	105
	Total	124	287	28	13	10	28	1	491
Commercial fishing	1980	22	21	7	0	7	0	0	57
	1981	91	9	3	2	0	0	0	105
	1982	14	42	23	2	11	9	1	102
	1983	0	27	0	0	1	0	0	28
	Total	127	99	33	4	19	9	1	292
NUSCo sampling	1980	18	38	1	0	0	0	0	57
	1981	21	5	0	7	0	0	0	33
	1982	12	155	9	1	0	8	0	185
	1983	0	129	0	2	1	1	0	133
	Total	51	327	10	10	1	9	0	408
Impingement at MNPS	1980	1	2	0	0	0	0	0	3
	1981	0	0	0	10	0	0	0	10
	1982	0	3	0	3	1	1	0	8
	1983	0	3	0	2	0	0	0	5
	Total	1	8	0	15	1	1	0	26
Miscellaneous[c]	1980	2	0	0	0	0	0	0	2
	1981	3	0	0	0	0	0	0	3
	1982	0	1	3	0	1	0	0	5
	1983	0	0	0	0	0	0	0	0
	Total	5	1	3	0	1	0	0	10

[a] Includes various locations along shoreline west of Black Point to the Connecticut River.

[b] Year here and following refers to year in which fish were tagged. Number recaptured includes 390 released alive (mostly by NUSCo), 86 of which were caught again once, 5 twice, and 1 three times.

[c] Includes recaptures from the CT DEP, Project Oceanology, and unknown sources.

smaller at time of tagging than most of the fish sexed, they may have had greater initial mortality following tagging, shed tags at a greater rate than larger fish, or were less vulnerable to capture. Many of them were probably smaller females since their movements tended to be similar to that of mature females. Proportionately, more females and fish of undetermined sex were taken from distant locations than males. Since even small males were readily sexed during the spawning season, it is likely that most of them were identified when tagged.

Table 19. Months at liberty for disc-tagged winter flounder by method of recapture.

Months at liberty	Sport fishing	Commercial fishing	NUSCo sampling	Impingement at MNPS	Miscellaneous[a]	Total	Percent of total
0-3	172	67	213	20	6	478	39
4-6	86	80	47	4	2	219	18
7-12	102	55	91	2	1	251	20
13-18	74	46	21	0	1	142	12
19-24	37	33	28	0	0	98	8
25-40	19	11	8	0	0	38	3
Total	490[b]	292	408	26	10	1,226[b]	--

[a] Includes recaptures from the CT DEP, Project Oceanology, and unknown sources.

[b] Month of one recapture not known.

Most (70%) of the returns were from waters near MNPS (Table 20). Similar to the 1973-74 study, movement out of the area tended to be to the east as about three times as many recaptures occurred there in comparison to the west. Most fish were taken in Fishers Island and Block Island Sounds, but 23 winter flounder were taken in waters near Martha's Vineyard and on Nantucket Shoals. One fish was caught off Cape Cod in February 1983 and in February 1985 one specimen was caught on Georges Bank. This specimen plus the very large 8-yr old mentioned above indicated that a very small interchange may occur between inshore and offshore stocks. Howe and Coates (1975) reported that a few percent of the winter flounder they tagged in inshore waters of southern Massachusetts were recaptured on Georges Bank and vice versa. Georges Bank winter flounder are usually recognized as a distinct race (Perlmutter 1947; Lux et al. 1970).

Table 20. Location of recaptures of disc-tagged winter flounder from December 1980 through September 1986.

Recapture location	Tagging location: Niantic Bay	Niantic River	Twotree	MNPS Intakes	Bartlett Reef	Jordan Cove	Miscellaneous	Total
Local								
Niantic Bay	127	60	13	14	4	4	--	222
Niantic River	37	495	4	4	--	3	--	543
Twotree	2	--	12	2	--	--	--	16
MNPS Intakes	1	8	--	15	1	1	--	26
Bartlett Reef	--	--	--	1	2	--	--	3
Jordan Cove	5	17	--	--	--	25	--	47
East								
New London Co., CT	43	35	17	2	9	11	1	118
Suffolk Co., NY	6	17	4	--	2	--	--	29
Washington Co, RI	35	37	9	1	4	--	--	86
Newport Co., RI	1	--	--	--	--	--	--	1
Barnstable Co., MA	--	1	--	--	--	--	--	1
Dukes Co., MA	4	3	2	--	1	--	1	11
Nantucket Co., MA	3	7	1	--	1	--	--	12
Georges Bank	--	1	--	--	--	--	--	1
West								
New London Co., CT	10	5	1	1	1	1	--	19
Middlesex Co., CT	4	11	1	2	--	--	--	18
Suffolk Co., NY	10	6	2	--	2	--	--	20
New Haven, Co., CT	7	2	8	--	3	--	--	20
Fairfield Co., NY	2	1	--	--	--	1	--	4
Bronx Co., NY	--	--	--	--	1	--	--	1
Unknown[a]								
Connecticut	1	1	--	--	--	--	--	2
New York	1	--	--	--	--	--	--	1
Rhode Island	4	7	--	--	1	--	--	12
Massachusetts	5	7	--	--	--	1	--	13
Virginia[b]	--	1	--	--	--	--	--	1
Total	308	722	74	42	32	47	2	1227

[a] Mostly from fish markets.
[b] Most likely caught in Rhode Island.

In general, Niantic River and Bay winter flounder showed patterns of movement similar to those reported by others. Lobell (1939) noted that concentrations of winter flounder near Block Island in summer were fish from the Long Island region. Weber and Zawacki (1986) tagged and released winter flounder in two bays off western Long Island Sound, New York. Most of the recaptures were made locally and many of the distant returns were from areas to the east during summer and fall. However, less than 10% of the recaptures were made outside of Long Island Sound. The majority of the fish seemed to return to areas around the tagging site in subsequent years. Danila and Kennish (1981) and Scarlett (1986) reported that fish resident in central and northern New Jersey estuaries from fall through spring moved offshore and to the north and east for summer. Howe and Coates (1975) found that winter flounder south of Cape Cod tended to move offshore to the southeast when water temperatures exceeded 15 °C. The Niantic River winter flounder population appears to have some individuals that remain in local waters throughout the year, yet others are able to move relatively long distances and successfully return each winter before spawning.

Stock identification

The ability to identify and separate stocks is important when quantifying impacts. If more than one discrete population is affected, then losses may be partitioned accordingly. A study was undertaken for NUEL by the University of Rhode Island in 1980-81 to investigate techniques for differentiating winter flounder stocks using a specific biochemical technique (Schenck and Saila 1982). The method chosen was direct tissue isoelectric focusing of eye lens proteins. Briefly, when a constant electrical potential is applied to a pH gradient formed within a gel, each of the separate proteins migrates to its isoelectric point, where it has no net electrical charge. This allows for the differentiation of even closely related protein molecules and provides a criterion of genetic homogeneity. A brief summary of the study follows.

The first portion of the work in 1980 examined fish taken from major estuaries or embayments over a relatively large (125 km) geographical area. Fish collected from New Haven, the Connecticut River, the Niantic area, Thames River, Mystic River, and Charlestown Pond separated well when the data concerning the presence or absence of certain proteins were used with linear discriminant analysis. Classification of individual winter flounder was highest in the correct area of capture with misclassification tailing off as a function of geographical distance from the source of each fish.

The 1981 study focused on a much smaller geographical area (23 km) using fish from the Connecticut River, Niantic River, Niantic Bay, Jordan Cove, and the Thames River. These areas were thought to be likely sources of winter flounder affected by MNPS operations. Because of the smaller area in which fish were taken, the samples were much more homogeneous than the ones in 1980. Fish from Jordan Cove often misclassified into Niantic River or Bay or the Thames River. The Connecticut River winter flounder also misclassified frequently. Upon further examination, most of the latter fish were found to have been sexually immature, which was thought to have caused the failures in discrimination. The seasonal samples taken in Niantic Bay over the year showed substantial variability; summer and fall samples were not significantly different, but other seasons were.

The major conclusion of the study was that the technique was able to distinguish between stocks or subpopulations of winter flounder found about 5 to 10 km apart. Winter flounder in the area around MNPS appeared to form separate stocks only during the winter and spring spawning season, with intermixing greatest during summer and fall. The areas immediately in the vicinity of the plant (Niantic River and Bay and Jordan Cove) appeared to be inhabited by substocks that intermix significantly. Additional interchange took place with stocks from more distant areas, such as the Connecticut and Thames Rivers. Immature fish were impossible to classify using the techniques and analyses employed during the study. They were a heterogeneous group either because they were well-mixed over the geographical range of the study or because juveniles showed a lack of differentiation for the particular proteins examined. As young-of-the-year fish were not examined, this conclusion may not apply to them.

The findings of this study along with tagging data indicated that at certain times of the year the winter flounder impinged at MNPS as well as throughout the study area were a mixture of a number of different spawning stocks. Consequently, the long-term effects of this particular impact on the Niantic River stock are somewhat reduced because of the dilution. Furthermore, the degree of intermixing implies that interchanges frequently occur among local stocks; this would also help to mitigate losses particular to any one of them.

Larval studies

Several special studies and analyses have been conducted to identify possible sampling biases in the larval winter flounder data base. Because of the biases there are limitations in the usefulness of some of the data. Therefore, the results of these special studies are presented first to identify these limitations.

Net extrusion studies

The effects of mesh size and tow duration on the sample density of larval winter flounder were examined in the field and laboratory. Field comparisons of the collection of 1-mm size-classes were made for 333 and 505-µm mesh nets from the 1974 and 1975 data (Fig. 8). Based on paired comparisons, the

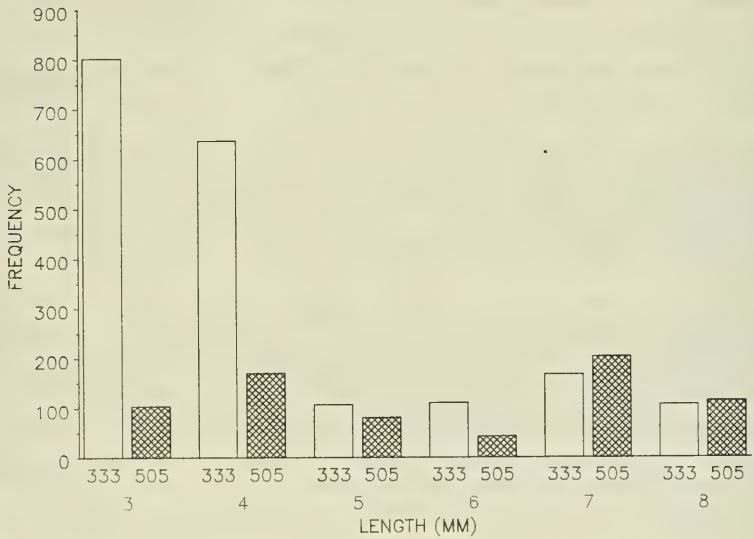


Figure 8. Comparison of larval winter flounder length frequencies by 1-mm size-class collected in paired 333- and 505-µm mesh nets during 1974 and 1975.

333-μm mesh collected significantly greater densities of the 3- and 4-mm size-classes with no differences detected for the other size classes. The effect of mesh size (202- and 333-μm) and tow duration (6- and 15-min) on net extrusion of early developmental stages (Stage 1 and 2) was examined in 1984 during February and March (NUSCo 1985). In the comparison between the two mesh sizes, Stage 1 larvae were collected in greater densities with 202-μm mesh in 21 of 28 paired comparisons, which represented a significant difference. No difference was found for Stage 2 larvae, with their density in the 202-μm mesh greater in only 13 of the 28 comparisons. No difference in the collection density of Stage 1 and 2 larvae was found between the 6- and 15-min tow durations (16 comparisons). The results of laboratory comparisons of 202- 333-, and 505-μm mesh (NUSCo 1986a) verified the findings of the field studies with the greatest retention in 202 μm (92%), followed by 333 μm (78%), and 505 μm (63%).

Prior to 1984, all ichthyoplankton sampling at MNPS was conducted with 333- or 505-μm mesh nets and during this period the early developmental stages were probably undersampled. This undersampling limited the use of these data to examine the abundance and distribution of early developmental stages. Although only a 333-μm mesh net was used for all recent entrainment collections, it probably did not result in underestimates because 202-μm mesh nets were used at station NB during 1985 and very few Stage 1 larvae were found in Niantic Bay (NUSCo 1986a).

Diel behavior

Diel behavior patterns of larval winter flounder could affect sample densities and bias abundance estimates. A comparison of day and night collections was made at three stations where balanced day and night sampling was conducted during several years: EN from 1976-85, NB from 1979-85, and C from 1980-83. The percentages that each 1-mm size-class made up of day and night samples were examined (Fig. 9). At stations EN and NB an increasing percentage of the 5- to 6-mm and larger size-classes occurred in night collections. Although the 5-mm and larger size-classes were more prevalent in night samples at station C in the Niantic River, the difference was less apparent than found at the other two stations. Comparison of day and night collections by developmental stage was possible with 1983 data (Fig. 10). Only in this year were all three Niantic River stations (A, B, and C) sampled during both day and night throughout the season. Stage 4 larvae at station A and Stage 1 larvae at stations EN and NB were excluded because of low collection densities. No consistent difference between day and night

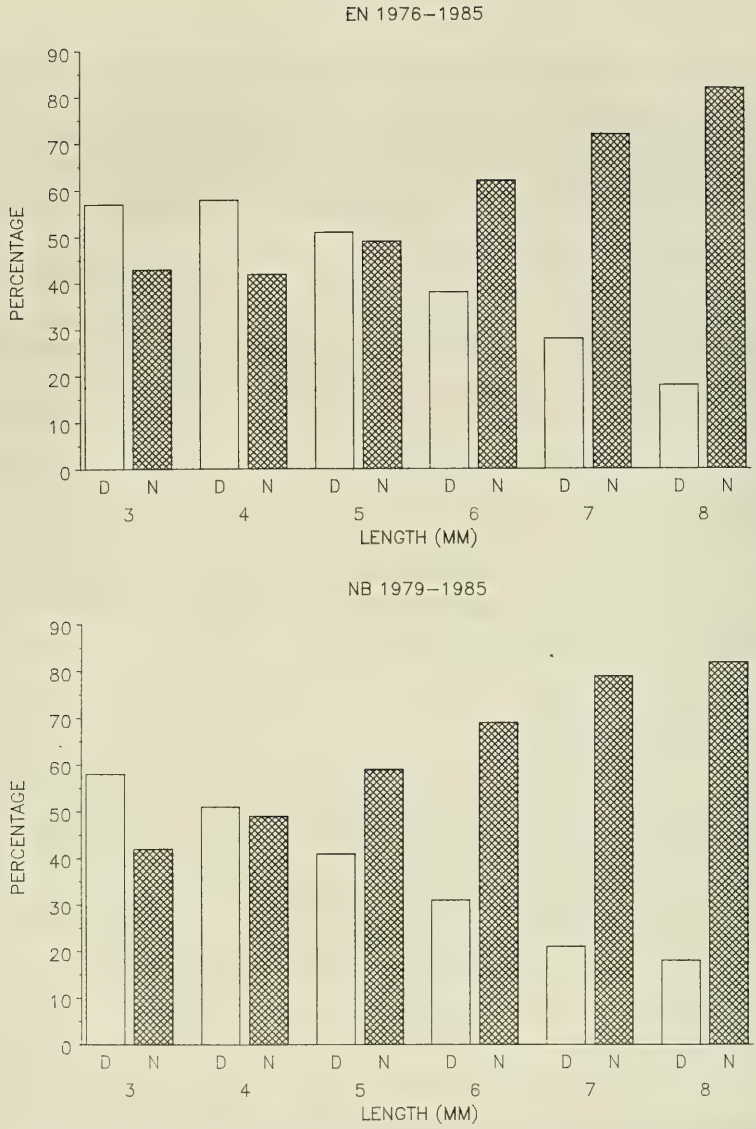
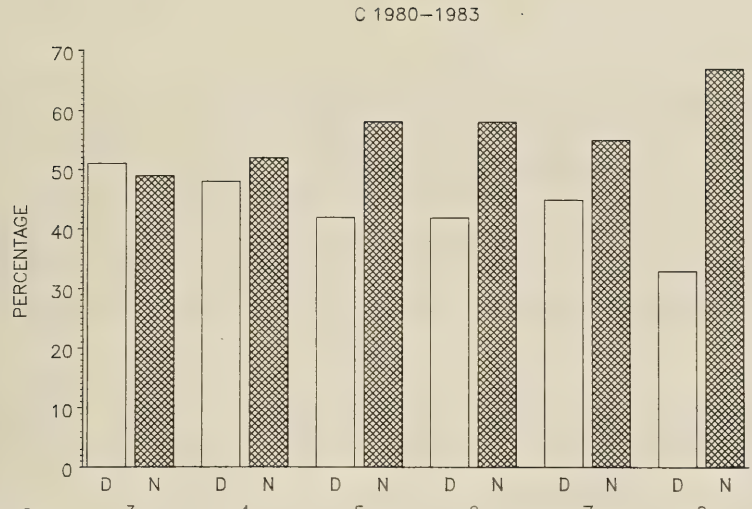


Figure 9. Comparison of the percentage of larval winter flounder takem in day (D) and night (N) collections by 1-mm size-class at stations EN, NB, and C during various time periods.



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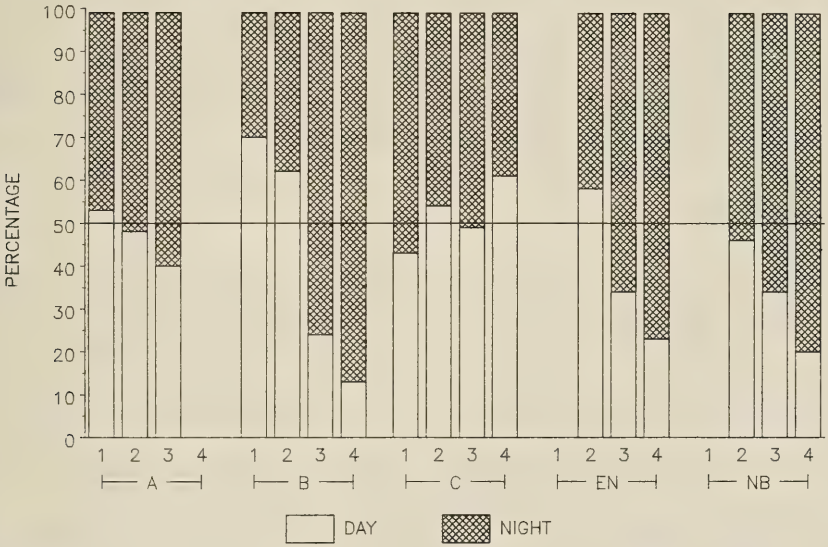


Figure 10. Comparison of the percentage of larval winter flounder taken in day and night collections by developmental stage at each station in 1983.

collections were found for Stage 1 and 2. For Stages 3 and 4, percentages were noticeably higher during the night at stations B, NB, and EN, but not at C. These diel differences have been attributed to vertical movement of larvae from on or near bottom into the water column at night (NUSCo 1984). This behavior pattern appears during Stage 3 following development of fin rays. The predominant length range of Stage 3 larvae is 5.0 to 7.5 mm (NUSCo 1984, 1985, 1986a) and the behavior pattern found for Stage 3 and 4 larvae agreed with the results based on size-classes. The lack of diel vertical movement of Stage 3 and 4 larvae at station C suggested that other factors in the lower river, such as tidal currents must have affected their behavior.

Comparison of day and night collections showed that abundance estimates based on daylight collections could underestimate the abundance of 5 mm and larger larvae. This was the reason for reducing sampling in the Niantic River to only night during May and June starting in 1984. Entrainment sampling remained balanced between day and night for the estimation of entrainment. Sampling at station NB remained balanced because this station was also used to monitor other species. The daylight sampling bias severely limited the usefulness of earlier data in examining the abundance and distribution of larval winter flounder. In 1974 and 1975, collections were only made at night once a month and from 1976-78 and none were made during the larval winter flounder season. In addition, the special larval winter flounder sampling in the Niantic River during 1979 consisted of only daylight collections.

24-h studies

Sampling was conducted over 24-h periods at station C to examine the effect of tidal stage on the sample density of larval winter flounder (Fig. 11). Two studies in 1983 occurred when the predominant developmental stages were 3 (62%) and 4 (30%). During 1984, when two additional studies were conducted, Stage 1 (48%) and 2 (51%) dominated. There was an apparent tidal effect on sample densities in both studies in 1983 and the March 19 study in 1984. Harmonic regression was used with log-transformed data to relate changes in density to tidal stage for these three studies (NUSCo 1984, 1985). A 12-h tidal period was used with slack low at hours 0, 12, and 24 and slack high at hours 6 and 18. Satisfactory fits were achieved for Stage 1 larvae ($R^2 = 0.58$) in the March 19, 1984 study and the combination of Stage 3 and 4 larvae on both sampling dates ($R^2 = 0.45$) in 1983. Analysis of covariance of the 1983 studies (NUSCo 1984), with tidal effect as described by the harmonic regression as the covariate, showed

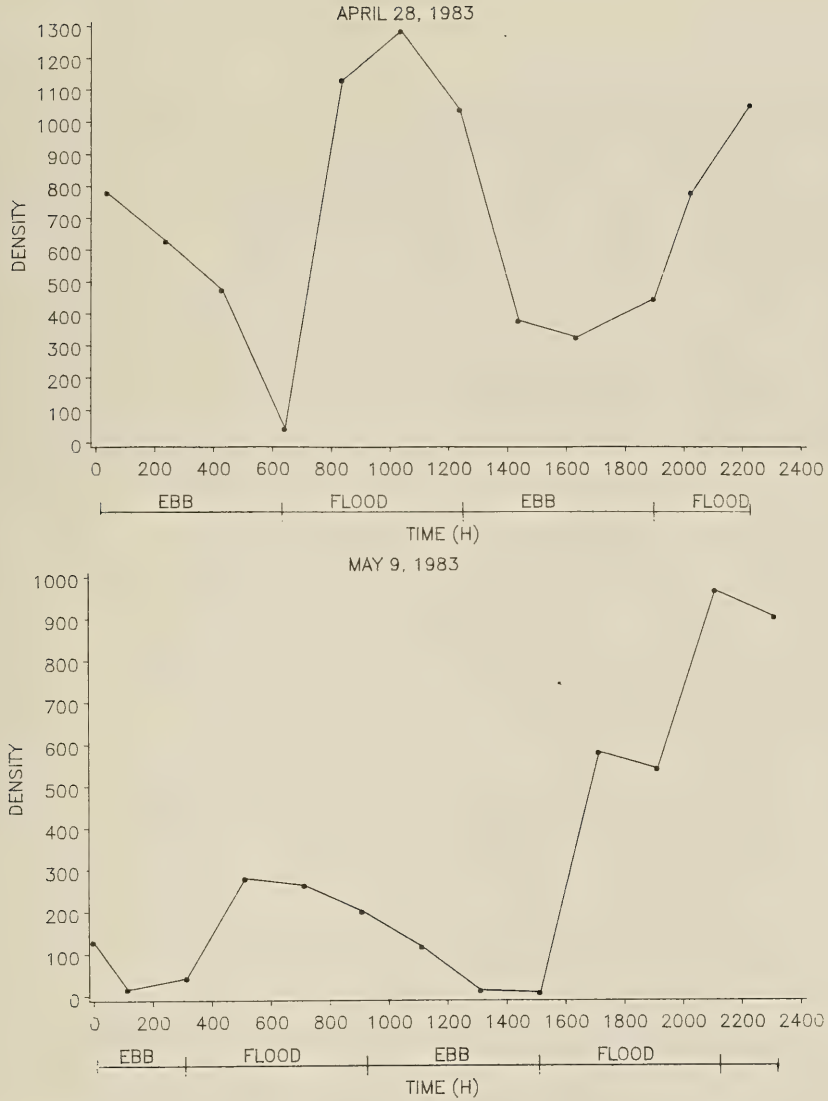


Figure 11. Larval winter flounder densities per 500 m^3 for 24-h studies at station C in 1983 and 1984 with time of collection and tidal stage.

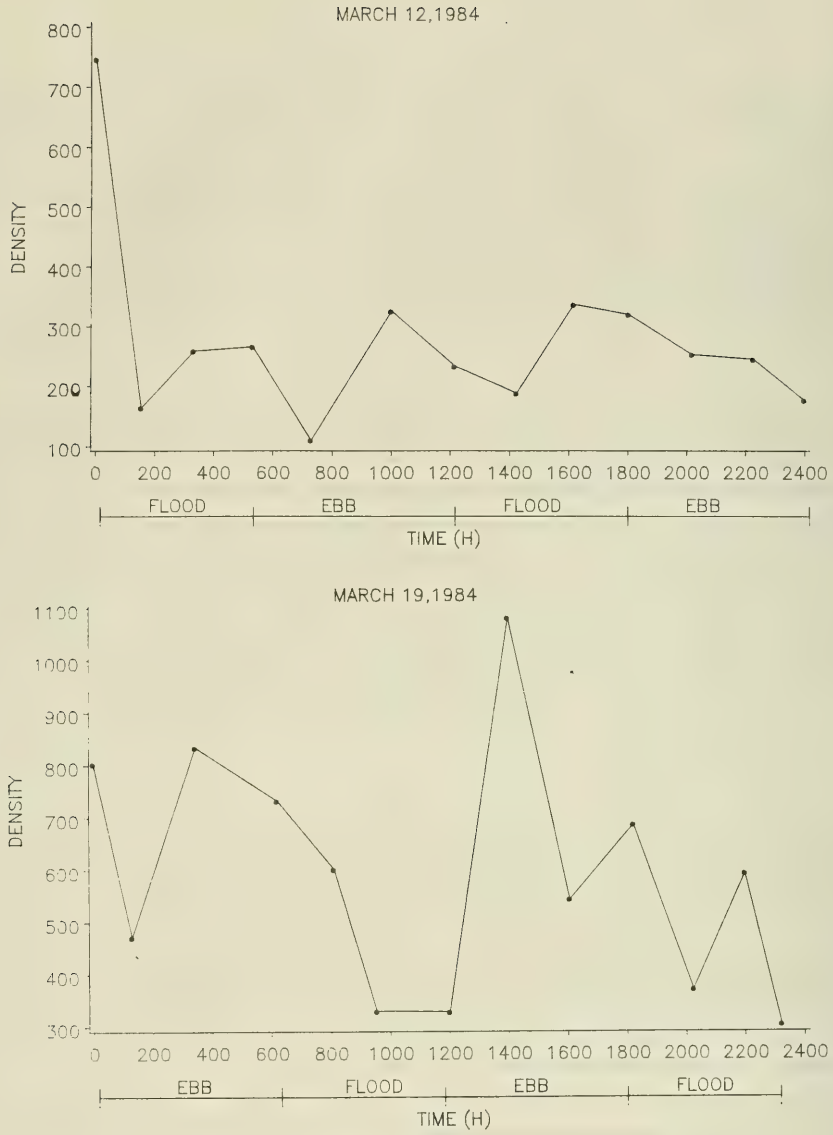


Figure 11. cont.

that an additional 19% of the total corrected sum of squares was due to the difference between the two sampling dates (April 28 and May 9). Based on the harmonic regressions, the effect of tidal stage on sample density at station C was inverse for Stage 1 compared to Stage 3 and 4 larvae (Fig. 12). Stage 1 density increased during ebb tides and declined during flood tides. This pattern was attributed to the flushing of Stage 1 larvae from the upper portion of the Niantic River, where they were more abundant, into the lower portion during ebb tide (NUSCo 1985). The decline during flood tide was caused by water from Niantic Bay that contained few Stage 1 larvae entering the river. The inverse relationship for Stage 3 and 4 larvae was attributed to vertical migration in response to tidal flow as a retention mechanism (NUSCo 1984), where the larvae remained on or near the bottom during an ebb tide and moved up in the water column during flood tides. This tidal behavior of Stage 3 and 4 larvae would help explain the lack of diel behavior at station C, as shown above (Fig. 10).

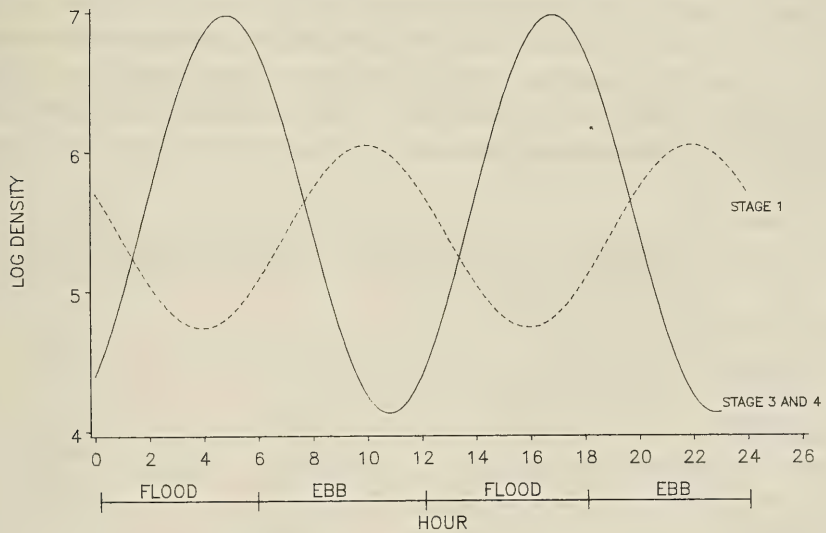


Figure 12. Harmonic regression models of larval winter flounder abundance (log density per 500 m^3) during 24-h tidal cycles at station C for developmental Stage 1 and Stage 3 and 4 combined.

Due to the effect of tides on sample density, starting in 1984 sampling time was based on tidal stage. During the occurrence of Stage 1 larvae sampling was conducted close to low slack and when Stage 3 and 4 larvae were common sampling was conducted during the latter portion of a flood tide. Prior to 1984, larval abundance in the lower portion of the Niantic River may have been underestimated, as sampling was not synchronized with tidal stage.

Abundance and distribution

The distribution of larval winter flounder near MNPS was examined based on the area-wide sampling conducted during 1974 and 1975, when up to 16 stations were sampled (Fig. 3). To reduce sampling bias the data were restricted to oblique samples collected with 333-μm mesh nets. The date of peak abundance was estimated from the point of inflection (Equation 8) of the Gompertz function to compare the temporal distribution in six areas. The grouping of stations resulted in 2 to 9 samples per week in 1974 and 2 to 15 samples per week in 1975 for calculating a weekly mean used to construct the cumulative density curve for the Gompertz function. Assuming that the Niantic River was the primary source of winter flounder larvae in the area, the progressive date of peak abundance would provide a relative measure of dispersal rate from the Niantic River. All fits of the Gompertz function exceeded an R^2 value of 0.97. A similar pattern of the time of peak abundance was found for both years (Table 21). As expected, the earliest

Table 21. Estimated date of peak abundance for larval winter flounder based on the inflection point of the Gompertz function for six areas around Millstone Point in 1974 and 1975.

Area	1974	1975
Niantic River	27 Feb	13 Mar
Mouth of the Niantic River	30 Mar	29 Mar
Mid-Niantic Bay	4 Apr	3 Apr
Jordan Cove	7 Apr	15 Apr
Twotree Channel	10 Apr	16 Apr
Offshore	13 Apr	17 Apr

peak was in the spawning area of the Niantic River. The date of peak abundance progressed to the mouth of the Niantic River, mid-Niantic Bay, and then throughout the Millstone area. It took from 21 d in 1975 to 36 d in 1974 for the peak to progress from the Niantic River to mid-Niantic Bay, but once in the bay dispersal was rapid to other areas. This progressive pattern of peak abundance as a relative measure of dispersal rate agreed with the findings of the larval dispersal model (Saila 1976). The Niantic River has high larval retention characteristics, but once larvae enter Niantic Bay they spread throughout the area.

A more detailed examination of the abundance and distribution of larval winter flounder in the Niantic River and Bay was possible with the data collected from 1981 through 1985. Abundance curves (Equation 7) were constructed from the Gompertz function (all R^2 values of the Gompertz function exceeded 0.95) for each year in the Niantic River (stations 1 and 2 combined for 1981 and 1982; A, B, and C for 1983 through 1985) and Niantic Bay (stations NB and EN combined). If collected, daylight samples during May and June were excluded because they would have underestimated larval abundance during this period. Larvae were most abundant in the river during 1982 and 1985 and the time of peak abundance for all years occurred during mid to the latter part of March (Fig. 13). In the bay, larvae were most abundant in 1982 and 1983, with similar numbers during the remaining 3 yr. The timing of peak abundance varied more than for the river, ranging from about the second week of April in 1985 to the first week of May in 1981. Except for 1982, there was no apparent relationship between annual abundance in the river and bay. For example, in 1985 larvae were most abundant in the river but were not abundant in the bay, and in 1983 larvae were most abundant in the bay but were not abundant in the river.

The abundance of each developmental stage was compared in the Niantic River and Bay from 1983 through 1985, the only years that larvae were classified in developmental stages. The α parmeter from the Gompertz function (Equation 6) was used as an index of abundance. This function fitted the data well with all R^2 values exceeding 0.97 and the 95% asymptotic confidence intervals for the α parameter were small (Table 22). However, less than 20 observations were used and the actual intervals may be larger because asymptotic theory generally requires large data sets to apply. The most noticeable difference in the river was the high abundance of Stage 1 and 2 larvae in 1985 compared to the previous 2 yr. This occurred despite decreasing estimates of egg production from 1983 through 1985 (Table 11). The low abundance of Stage 1 and possibly early Stage 2 larvae in 1983 was partly attributed to net extrusion through the 333-μm mesh nets (NUSCo 1985). The previously discussed high densities of larvae in 1985

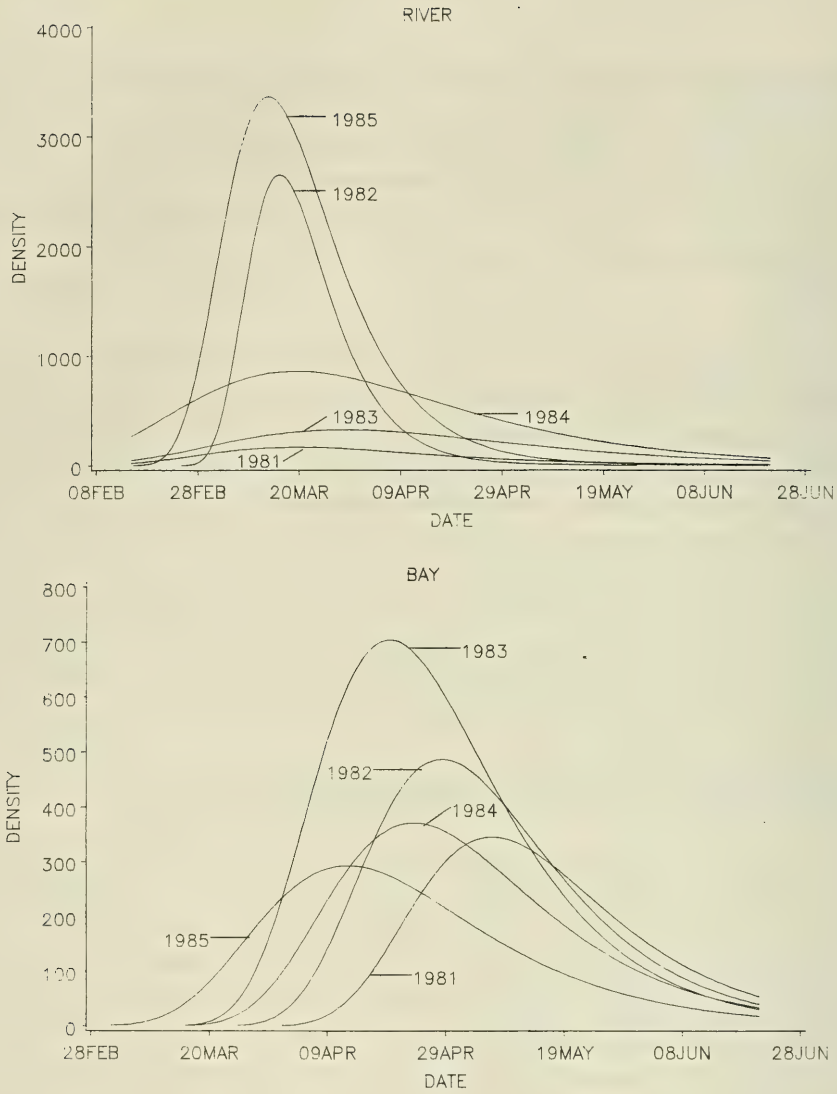


Figure 13. Estimated abundance curves (number/500 m^3) for larval winter flounder at Niantic River and Bay stations from 1981 through 1985.

in the river (Fig. 13) was a result of Stage 1 and 2 larval densities. In the river, densities of Stage 3 larvae were similar in 1983 and 1985, but both were lower than in 1984. The higher abundance of Stage 2 and the lower abundance of Stage 3 larvae in 1985 compared to 1984 indicated higher mortality during Stage 2 to Stage 3 development in 1985. The Gompertz function was not fitted to Stage 1 larvae in the bay because they were rarely collected there. Stage 2 abundance in the bay for 1984 and 1985 was much lower than in the river; this was expected because it is during this developmental stage that the larvae are tidally flushed from the river (NUSCo 1985). The low abundance of Stage 2 in the river compared to the bay in 1983 may be due to undersampling of early Stage 2 larvae in river because of net extrusion. The large decrease in abundance from Stage 3 to Stage 4 in the river and bay during all years was not completely attributed to mortality, but probably represented an undersampling of the older larvae. At Stage 4 of development, the left eye has migrated to or past the mid-line, the larvae become more demersal, and not as susceptible to capture with a plankton net. The undersampling of Stage 4 larvae should have remained constant from year to year; therefore, their decreasing frequency in the river and bay since 1983 probably represented a decrease in abundance.

Table 22. Larval winter flounder abundances and 95% asymptotic confidence intervals as estimated by the α parameter from the cumulative Gompertz function.

Developmental stage	1983	1984	1985
		Niantic River	
1	540 (509-570)	4075 (3727-4424)	7836 (7641-8031)
2	1350 (1317-1383)	2948 (2646-3250)	5783 (5621-5947)
3	1029 (1007-1051)	1846 (1667-2024)	1023 (965-1082)
4	306 (263-331)	270 (228-312)	162 (125-199)
		Niantic Bay	
2	1368 (1320-1417)	994 (961-1027)	944 (906-983)
3	2045 (1991-2099)	1326 (1272-1379)	1172 (111-1232)
4	620 (581-657)	355 (306-405)	127 (116-139)

The temporal occurrence of each developmental stage in the river and bay from 1983 through 1985 was compared using the dates of peak estimated abundance (Table 23). These dates were estimated from the inflection point (Equation 8) of the same Gompertz function that was used to estimate the α parameter above. Since 1983, the dates of peak larval abundance have been very similar for Stage 1 in the river and Stage 2 in the river and bay. The lag in peak abundance of Stage 2 in the bay compared to the river ranged from 17 to 23 d. This difference between the river and bay may be related to flushing rate because the average retention time of a passive particle in the Niantic River was reported as 25 d by Moore and Marshall (1967) and 27 d by Kollmeyer (1972). Within each year, the dates of peak abundance for Stage 3 were similar in the river and bay. The greatest difference in dates among years was for Stage 4 larvae, which in 1983 peaked earlier than either 1984 or 1985. The similarity in the estimated dates of peak abundance, particularly Stage 1 larvae in the river, indicated that peak spawning occurred approximately at the same time during the 3-yr period. Based on water temperatures of 2 to 3 °C during the latter portion of February and egg incubation times reported by Buckley (1982), peak spawning probably occurred in mid-February. The lack of Stage 1 larvae in the bay showed that spawning took place almost exclusively in the river and the lag in Stage 2 abundance represented the gradual flushing of larvae from the river to the bay. The similarity in the date of peak abundance for Stage 3 larvae between the the river and the bay in each year indicated that by this stage of development the dispersion of larvae from the river to the bay was completed.

Table 23. Estimated date of peak abundance of larval winter flounder in the Niantic River and Bay from 1983 through 1986.

Developmental stage	River			Bay		
	1983	1984	1985	1983	1984	1985
1	5 Mar	9 Mar	11 Mar	---	---	---
2	16 Mar	18 Mar	16 Mar	8 Apr	9 Apr	2 Apr
3	18 Apr	26 Apr	25 Apr	23 Apr	30 Apr	24 Apr
4	30 Apr	19 May	17 May	10 May	23 May	18 May

A comparison of the spatial distribution of each developmental stage in 1983 through 1985 was based on the cumulative abundance of weekly mean densities at each station (Fig. 14). As previously stated, Stage 1 larvae were collected almost exclusively in the river and their low abundance in 1983 compared to Stage 2 larvae was the reason for conducting net extrusion studies in 1984 and 1985. Their annual spatial distribution varied with the greatest numbers collected at stations B and C in 1984 and at station A in 1985. A similar pattern was found for Stage 2 larvae in 1984 and 1985. The previously mentioned high mortality from Stage 2 or 3 in 1985 was apparent at all river stations. Stage 3 and 4 larvae were primarily collected in the lower portion of the river at C and at Niantic Bay stations, but rarely at A. The decline in larval abundance since 1983 in the bay (Fig. 13) was reflected in decreasing frequency of Stage 3 and 4 larvae at EN and NB. The decline in Stage 4 larvae in the river and bay (Table 22) from 1983 through 1985 was evident at stations B, C, EN, and NB.

Time-based harmonic regression models were used to describe the fluctuations in abundance of larval winter flounder at station EN (1976-85) and NB (1979-85). Both models had a seasonal component and 1-yr and 6-mo terms (Table 24). It was possible to correct for autocorrelation at station EN because the data were evenly spaced in time. No long-term trends were present in either model. The models fitted the data well with R^2 values of 0.93 for EN and 0.91 for NB. The time-based models will be used as base-line information on the abundance of larval winter flounder for a comparison to their abundance during Unit 3 operation.

Table 24. Summary of time-based regression models to describe the occurrence of larval winter flounder at stations EN and NB.

Station	Model[a]	Model R^2
EN	$S\ \{\sin(1Y) - \cos(1Y) - \sin(6m) - \cos(6m)\} + A1 + A2 - A3 - A8$	0.93
NB	$S\ \{\sin(1Y) - \cos(1Y) - \sin(6m) - \cos(6m)\}$	0.91

[a] S = Season
nY, nM = period in years or months
An = autoregression coefficients

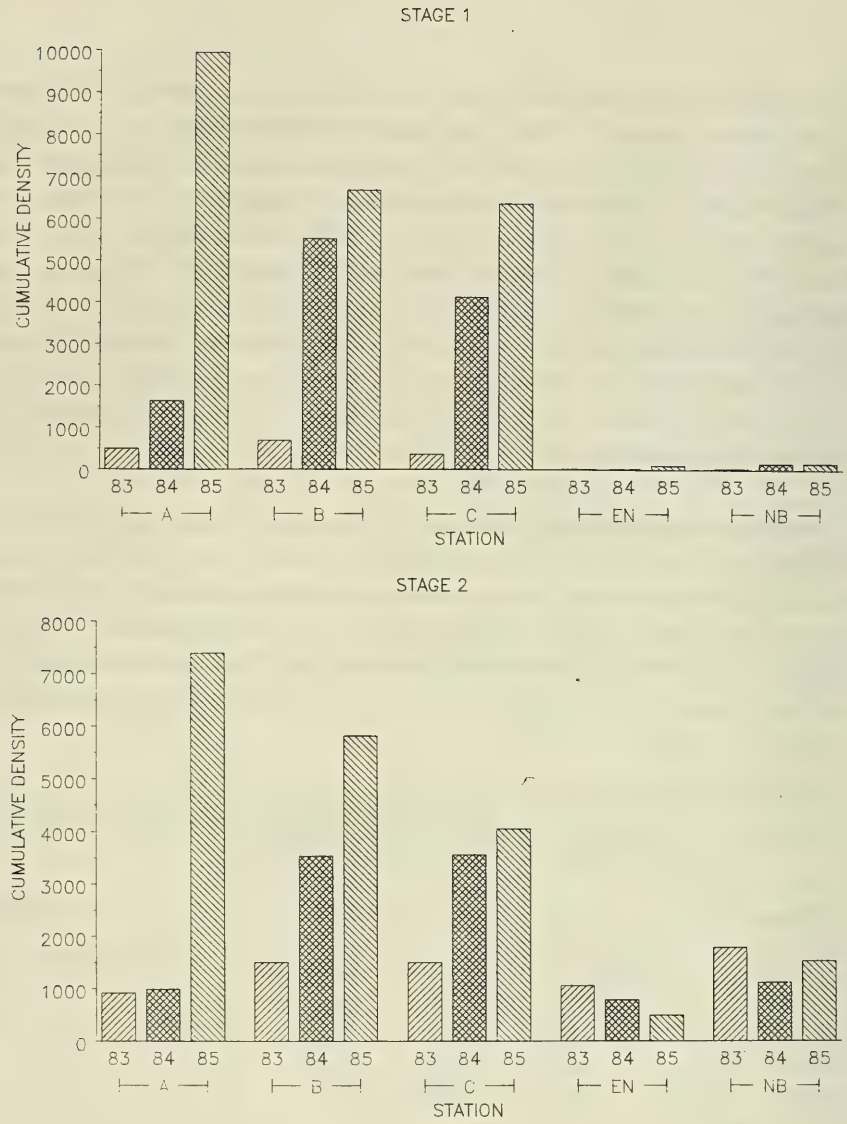
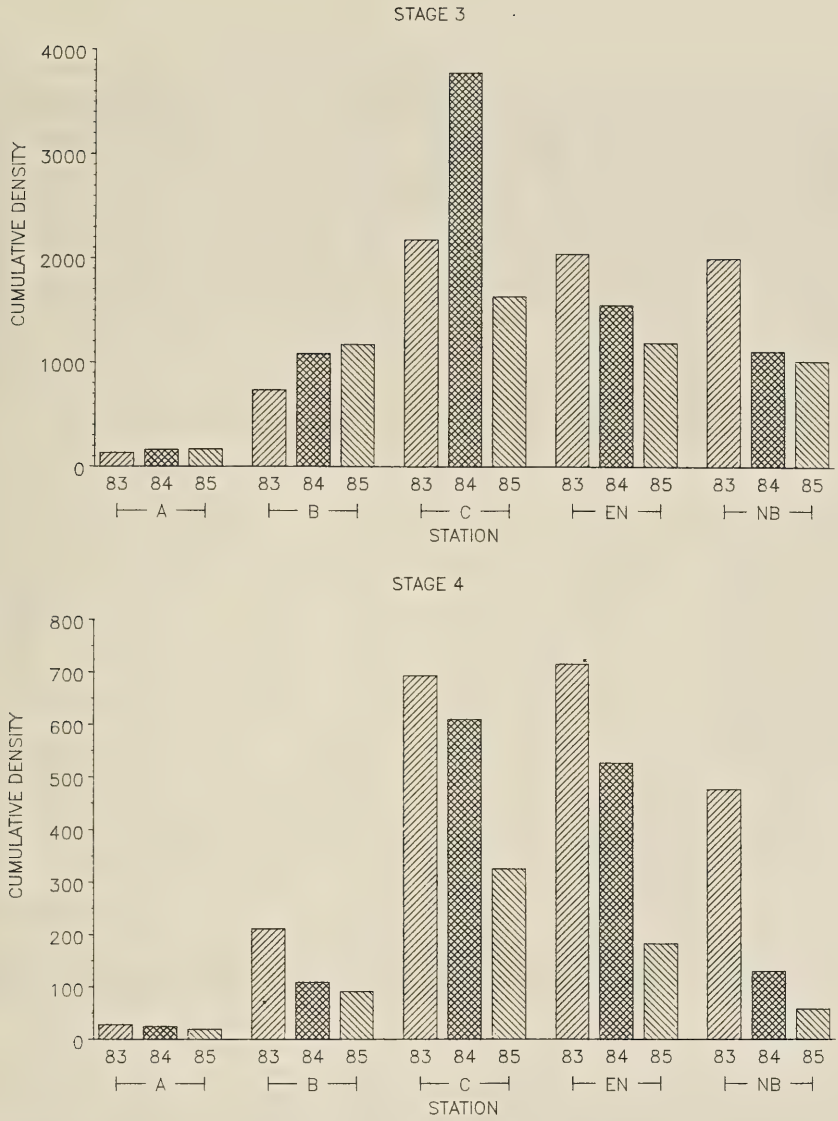


Figure 14. Cumulative density by developmental stage for larval winter flounder at each station from 1983 through 1985.



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The abundance and distribution of larval winter flounder in the Niantic River could be affected by predation. During their occurrence, the medusal stage of the lion's mane jellyfish (*Cyanea* sp.) was examined as a potential predator (Miller et al. 1986). Larval predation by these medusae has been observed in laboratory studies at NUEL. A medusa was placed in a container with laboratory-reared Stage 2 larvae. All larvae that came in contact with a tentacle were immediately stunned and even larvae that were not consumed by the medusa sank to the bottom and died. During the larval winter flounder season an examination of medusae in the Niantic River showed that up to 50% of the feeding jellyfish had fish larvae in their gastrovascular cavity. Medusae were primarily collected in the upper portion of the river at station A. Marshall and Hicks (1962) also found that jellyfish were most abundant in the upper river. During the 3-yr period there were noticeable differences in the abundance of medusae at station A (Fig. 15). Fewest were found in 1985, which corresponded to highest larval densities (Fig. 14).

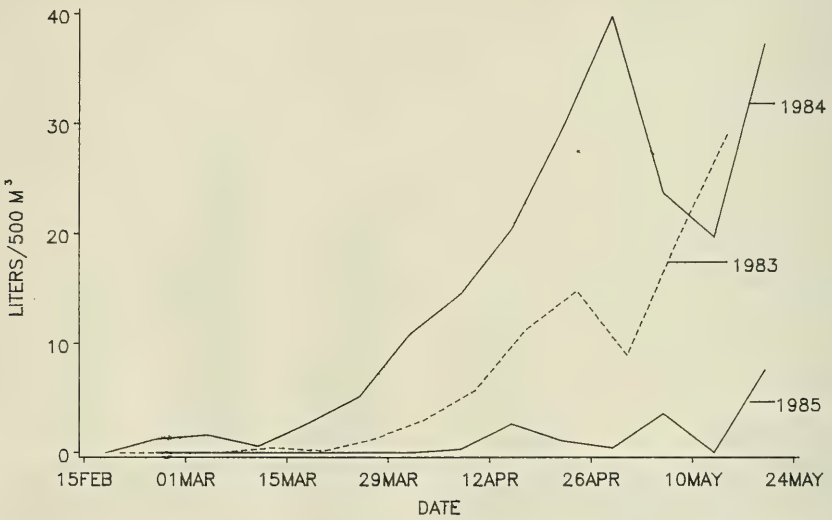


Figure 15. Weekly mean volume (liters/500 m^3) of *Cyanea* sp. medusae collected at station A in the Niantic River from 1983 through 1985.

Weekly densities of medusae at station A were estimated by comparing the volumes of medusae in a tow to the mean weekly medusoid bell diameter, using a medusoid diameter-to-volume relationship (Fig. 16). Density estimates in 1983 and 1984 reached a maximum of approximately 3 to 4 per m^3 but during 1985 never exceeded 1 per m^3 (Fig. 17). Considering the high jellyfish densities in 1983 and 1984 and with tentacles extending up to 10 to 15 cm below the bell, there was a relatively high probability that a larvae would come into contact with a medusa.

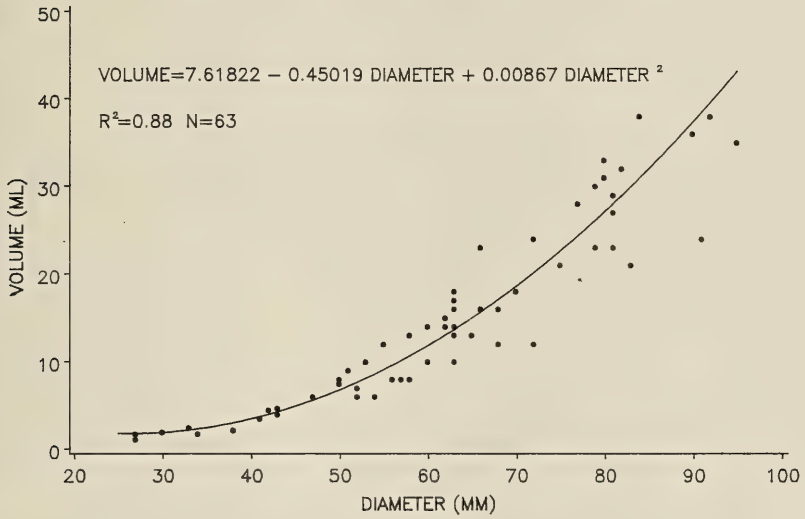


Figure 16. Polynomial regression of individual *Cyanea* sp. medusoid diameter to volume.

There are numerous accounts that jellyfish are predators of fish larvae. Several species of hydromedusae and the scyphomedusa *Aurelia aurita* were found to prey upon herring larvae (*Clupea harengus*) (Arai and Hay 1982; Moller 1984). Laboratory studies with cod (*Gadus morhua*), plaice (*Pleuronectes platessa*), and herring showed that the capture success by *A. aurelia* increased with medusa size (Bailey and Batty 1984).

Evidence of a causal predator-prey relationship on larvae of two European flatfish (*Pleuronectes platessa* and *Platichthys flesus*) by *A. aurita* and the ctenophore *Pleurobrachia pileus* was reported by van der Veer (1985). Pearcy (1962) stated that *Sarsia tubulosa* medusae were important predators of larval winter flounder in the Mystic River, CT, and had greatest impact on younger, less motile individuals. Crawford and Carey (1985) reported large numbers of the moon jelly (*A. aurata*) in Point Judith Pond, RI and felt that they were a significant predator of the pelagic larval stage of winter flounder. Although no causal predator-prey relationship in the Niantic River was established, there was strong circumstantial evidence that the lion's mane jellyfish was an important source of mortality for winter flounder larvae.

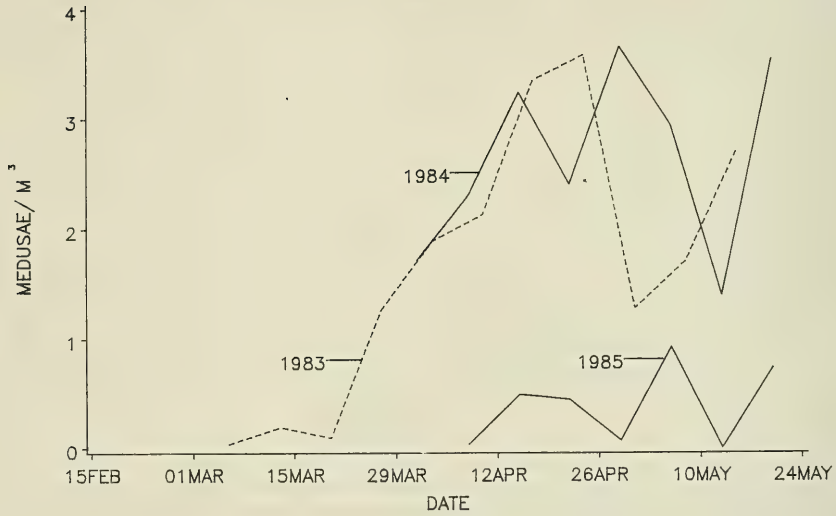


Figure 17. Estimated abundance (number/m^3) of *Cyanea* sp. medusae based on the volume to mean diameter relationship.

Age and growth

Otoliths from larval winter flounder were examined to determine if an age-length key could be constructed as deposition of daily increments on otoliths has been reported for many fish (e.g., Panella 1971, 1974; Brothers et al. 1976; Laroche et al. 1982; Campana and Neilson 1985). Radtke and Scherer (1981) reported that winter flounder larvae deposited a daily increment on otoliths following yolk absorption. In 1984, otoliths from 104 larvae collected in the Niantic River were examined and 81 of these were sufficiently clear to count the number of increments. Total increment counts ranged from 0 to 1 for Stage 1, 0 to 6 for Stage 2, 5 to 35 for Stage 3, and 17 to 50 for Stage 4 (Fig. 18). Based on the low counts for Stage 2 and older larvae, daily increment deposition was not apparent. The otoliths of known-age

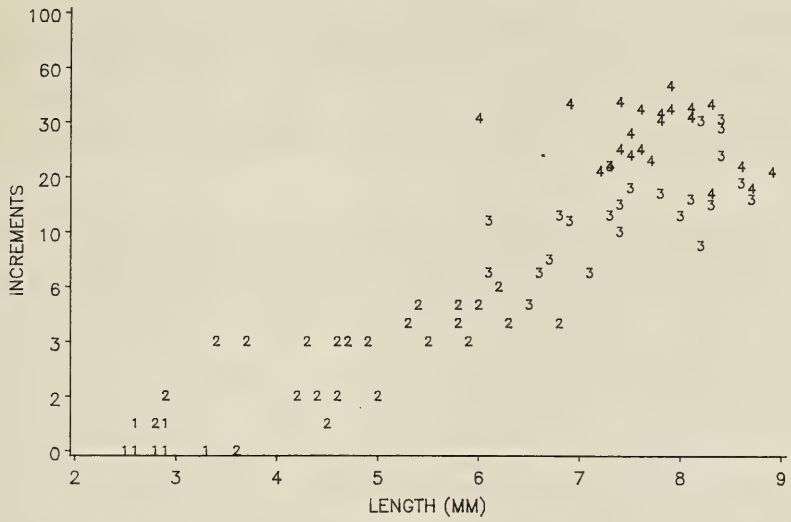


Figure 18. Otolith increment count by length for each developmental stage of larval winter flounder from 1984 samples.

larvae reared in the laboratory in 1985 were examined to determine if daily increments were formed (Table 25). Increment formation did not start until after yolk absorption, which agreed with the findings of Radtke and Scherer (1981). But contrary to their findings, daily increment deposition was not evident when otoliths were examined with a light microscope using transmitted light. A comparison of the number of increments on known-age laboratory-reared larvae by length agreed well with increment counts from the 1984 field data. Campana and Neilson (1985) stated that daily deposition may occur, but due to the resolution limit of a light microscope individual increments cannot be seen. Ongoing research at the University of Rhode Island (Dr. A. Durbin, University of Rhode Island, Narragansett, RI, pers. comm.) has shown that daily increments on winter flounder otoliths were not discernible using a microscope with transmitted light, but increment definition could be enhanced with special grinding, polishing, coating, and reflected light techniques.

Table 25. Number of visible otolith increments from known age laboratory-reared larvae with the number and length range of individuals examined.

Age (days from hatching)	Increments	Number examined	Length range (mm)
7	0	3	3.6 - 3.7
11	0 - 2	3	3.8 - 4.4
21	3	1	4.5
28	2 - 3	2	5.0 - 5.8
35	3	3	5.1 - 6.1
42	4 - 5	3	5.8 - 6.9
49	6	1	7.2
56	7	2	6.6 - 7.8
70	30 - 32	3	6.3 - 7.1

The laboratory-reared larvae in 1985 were also used to examine developmental time and the effects of starvation on growth and development. The larvae were held in three aquaria and those in one were not fed. Water temperature ranged from 4.3 to 9.1 °C with a gradual increase occurring during the holding period. Mean length at hatching was 2.94 mm (SE = 0.017; n = 160). Yolk absorption occurred 10 d after

hatching and growth was similar in both fed and unfed treatments (Fig. 19). By 18 d after hatch most larvae in the unfed treatment were dead. Buckley (1980) reported 100% mortality at 6 to 9 d after yolk absorption for unfed laboratory-reared winter flounder larvae. From laboratory information it appeared that if no food was available within 8 d following yolk absorption, high mortality would occur. The starvation period may be even shorter due to the "point of no return" reported by Blaxter and Hempel (1963) and discussed by May (1974), as a starved larva will become too weak to feed and survive even if food is provided.

In the laboratory, the duration of Stage 1 was 10 d, Stage 2 was 32 d, Stage 3 was 14 to 28 d, and Stage 4 was less than 14 d. The estimates of Stage 3 and 4 were ranges because the sampling of larvae occurred at 2-wk intervals due to low abundance. These developmental periods were within the range of estimates for Stage 1, 2, and 3 larvae from the 1983-85 field data (Table 26). The developmental times from the field data were estimated by modal progression (NUSCo 1985) using the number of days between peak abundance of successive developmental stages. However, Hairston and Twombly (1985) demonstrated that changes in mortality rates could bias estimates of developmental time based on modal progression. Until the effects of mortality on the use of modal progression are determined, these estimates of developmental time from field data are of questionable validity.

Table 26. Estimated development time (days) for larval winter flounder Stages 1 to 3 based on modal progression for 1983 through 1985 from Niantic River collections.

Developmental stage	1983	1984	1985
1	11	9	5
2	32	39	40
3	12	23	22
Total	55	71	67

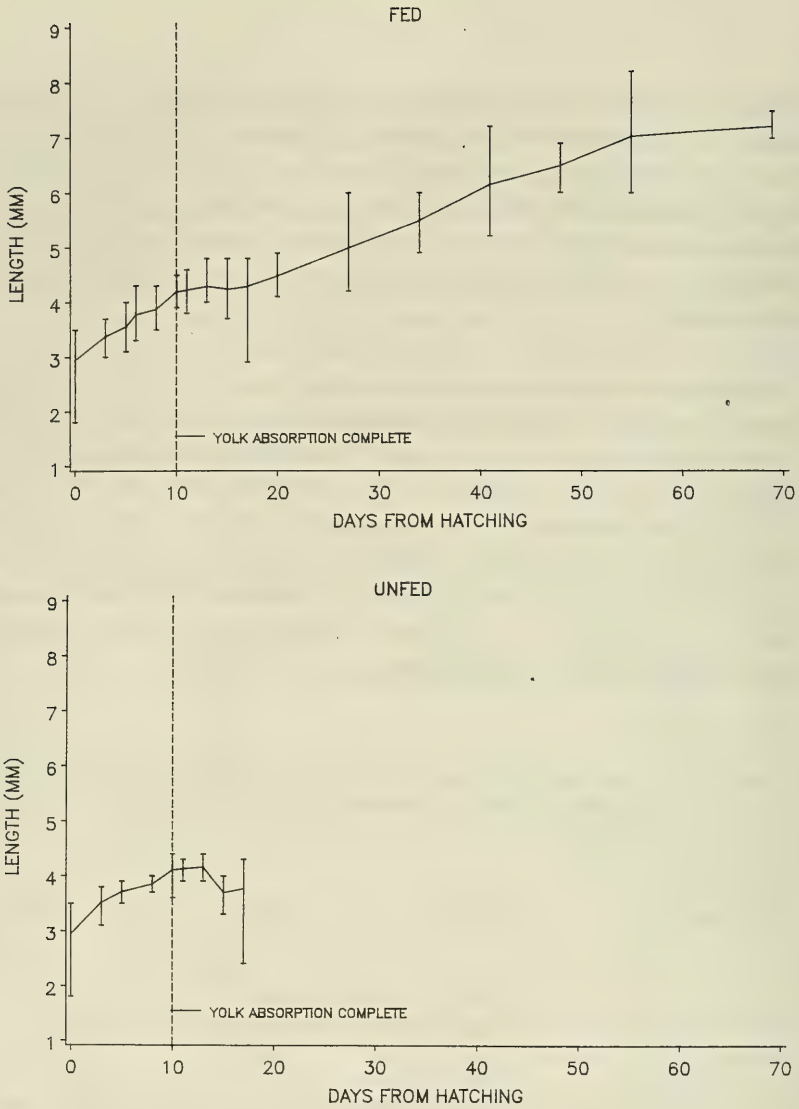


Figure 19. Mean length and range of laboratory-reared fed and unfed winter flounder larvae.

Examination of the length-frequency distribution of larvae collected from 1983 through 1985 showed a separation between the first three developmental stages by predominant 0.5-mm size-classes (Fig. 20). Stage 1 larvae were primarily in the 2.5 to 3.0 size-classes (84%), Stage 2 were 3.0 to 4.5 (88%), Stage 3 were 5.0 to 7.5 (87%), and Stage 4 were 6.5 to 8.0 (83%). These predominant size-classes for each developmental stage were similar in each of the years (NUSCo 1984, 1985, 1986a), indicating that stage development and length were closely related. Due to this relationship, larval developmental stages can be estimated from length measurements for data collected prior to larval classification by stage in 1983.

A comparison was made of the length-frequency distribution of larvae collected in the Niantic River and Bay during 1981-85 (Fig. 21). Like the spatial distribution of developmental stages (Fig. 14), smaller larvae dominated in the river and larger larvae in the bay. The 3.0-mm and smaller size-classes comprised over 50% of the larvae collected in the river during the 5-yr period, even though the collections in 3 of the 5 yr were made with 333-μm mesh nets and many of the smaller larvae were undersampled because of net extrusion. Based on the large decline from the 3.0- to the 4.0-mm size-class, highest mortality probably occurred at that size. Larvae in this length range were a combination of Stage 1 during yolk absorption and Stage 2 at first feeding. This apparent time of high mortality may represent the larval winter flounder "critical period", a concept first hypothesized by Hjort (1926) and discussed by May (1974) for marine fishes. They suggested that starvation may be a compensatory factor. This period of high mortality was not as evident in the catch curve of winter flounder larvae presented by Pearcy (1962) for the Mystic River. But the dominant size-class in his catch curve was 3.5-mm and his use of a 363 − μm net may have resulted in an undersampling of smaller larvae. The slight increase in percentage starting with the 6.0-mm size-class in the river and bay probably represented decreased growth in length for Stage 3 and 4 larvae during metamorphosis with a concurrent increase in body depth. Laroche (1981) reported that for winter flounder the percentage of body depth at the pectoral fin base to standard length increased from 9% for yolk-sac larvae to 31% for transformed larvae. The increasing percentage of larvae from smaller to larger size-classes in the bay again indicated that spawning primarily occurred in the river and larvae were gradually flushed into the bay. In the bay over 50% of the larvae collected were in the 5.0-mm and larger size-classes. Based on the length frequency-developmental stage relationship (Fig. 20), they were mostly Stage 3 and 4 larvae. The decline in percentage at about 6.5 to 7.0 mm in the river and bay represented the transition to demersal juveniles that were less susceptible to capture with a plankton net.

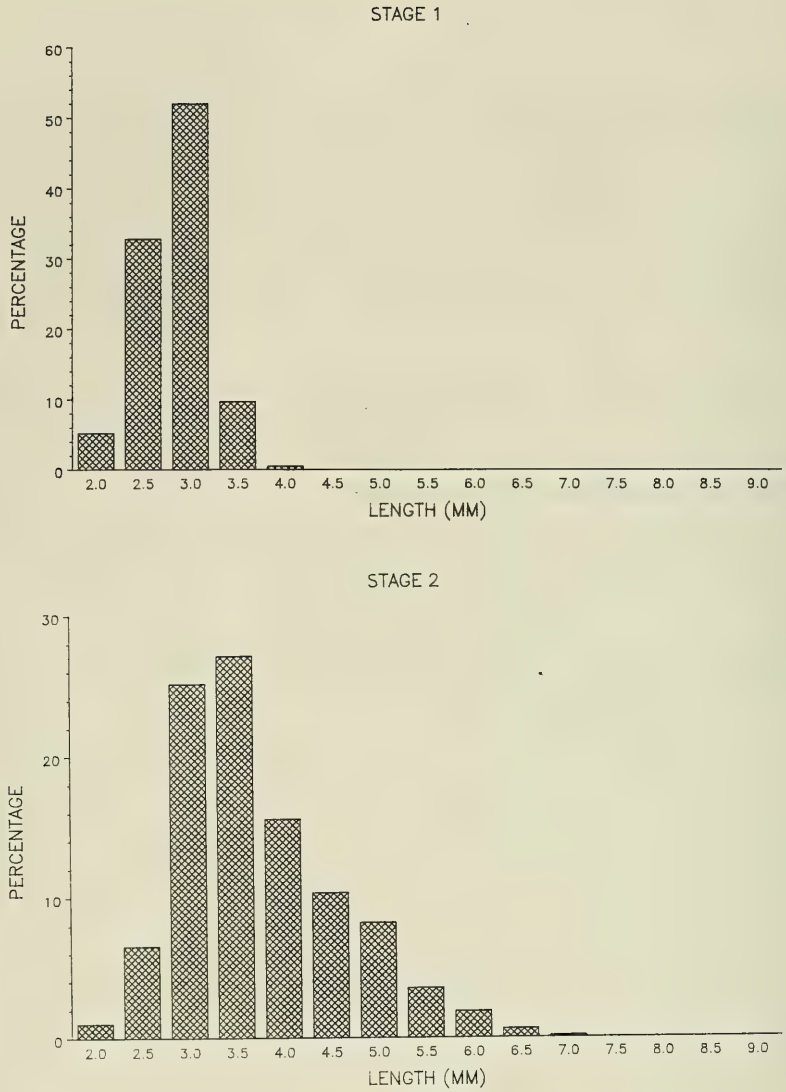
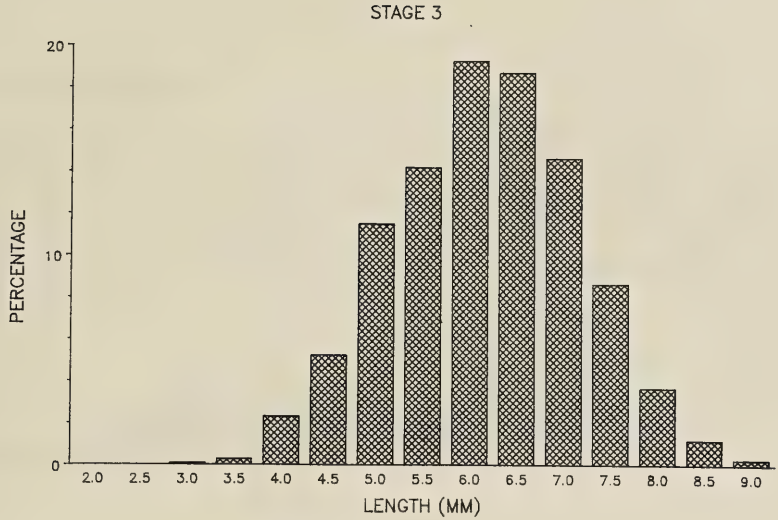
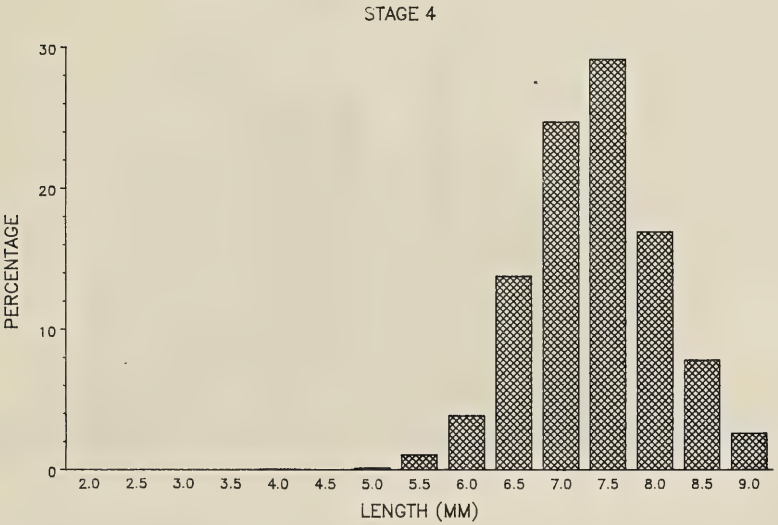


Figure 20. Length-frequency distribution of larval winter flounder by developmental stage for all stations combined from 1983 through 1985.





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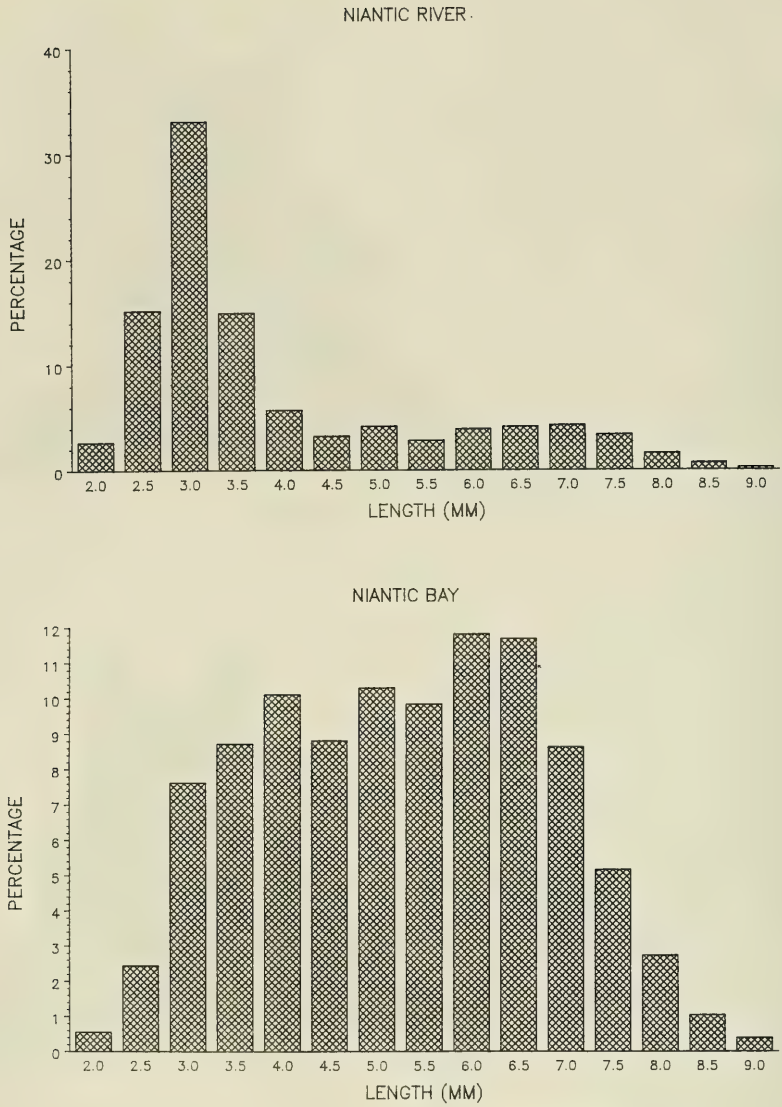


Figure 21. Length-frequency distribution of larval winter flounder for all stations combined in the Niantic River and Bay from 1981 through 1985.

Tidal export and import

The potential export or import of winter flounder larvae from or to the Niantic River was investigated in 1983 by sampling three ebb and flood tides at the time of maximum current velocity. Most of those collected were Stage 3 (45%) and 4 (48%) larvae. Many more larvae were collected during a flood tide (Fig. 22). This indicated that there was a net import of at least the later developmental stages into the river from the bay.

To quantify the net exchange of larvae between the river and bay, additional sampling at the mouth of the Niantic River was conducted in 1984 and 1985. Five tidal cycles were sampled at 1-h intervals.

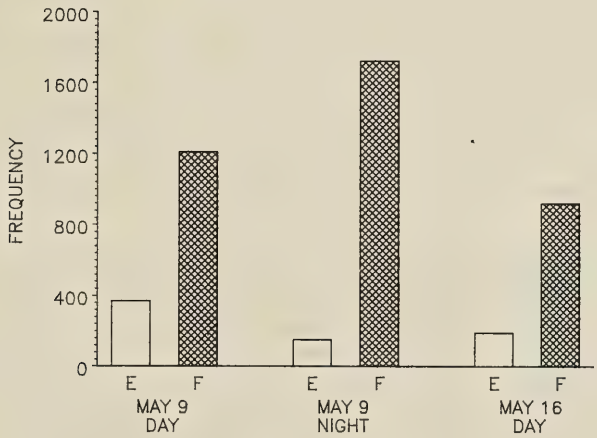


Figure 22. Frequency and collection times of larval winter flounder taken at the mouth of the Niantic River during maximum ebb (E) and flood (F) tidal currents.

The sampling dates were spaced over most of the larval winter flounder season to collect various developmental stages. In 1984, mostly Stage 1 and 2 larvae (92%) were collected on April 5 and on May 8 the larvae were primarily Stage 3 (85%). In 1985, all larvae collected on March 28 were Stage 1 and 2, on April 29 most were Stage 2 and 3 (99%), and on May 28 Stage 3 was dominant (89%). A few Stage 4 larvae were collected on two of the sampling dates (May 8, 1984 and May 28, 1985). Examination of the combined data from all five dates by percent occurrence of each developmental stage showed that Stage 1 and 2 larvae were more abundant during ebb tides and Stage 3 and 4 during flood tides (Fig. 23). Similarly, examination by size classes showed that larvae 4 mm and smaller were more abundant during an ebb tide and larvae 5 mm and larger were more abundant during a flood tide.

In order to determine if velocity measurements were comparable between ebb and flood tides, separate quadratic polynomial equations were fitted to hourly velocity measurements combined from each of the five ebb and flood tides sampled. Good fits were obtained with an R^2 value of 0.96 for both equations. The mean ebb duration was 6.8 h and flood duration was 5.8 h. The area under the curve for the flood tide (299.6) was smaller than for the ebb tide (397.7), indicating that flood velocities were low due to sampling location. To make ebb and flood velocities comparable, the flood velocities were estimated using a technique presented in NUSCo (1986a). The calculations of net exchange of larvae which follow were based on actual ebb current velocities and the adjusted flood current velocities.

Using data combined from the five sampling dates, net tidal exchange was estimated for each 1-mm size-class. The estimates were obtained by summing the number per 500 m^3 of larvae of each size-class in each hourly sample for the five sampling dates. The sum was multiplied by the estimated water velocity at the time of the hourly collection. This density-velocity adjustment accounted for changes in discharge volume during the tidal cycle. Because larvae collected during an ebb tide represented a loss from the river, the density-velocity value was made negative. A harmonic regression equation using a 12.6-h tidal cycle (the average duration of the five tides sampled) was fitted to density-velocity values. The area under the curve for each tidal stage was estimated by numerical integration of the regression equation using 5-min increments. Net tidal exchange was expressed as the percent return of a size-class on a flood tide compared to loss on a ebb tide (Table 27). The harmonic regression could not be fitted to the 2-mm size-class, because so few were collected on a flood tide. The results showed a net export of 4 mm and smaller size-classes and a net import of 5 mm and larger size-classes.

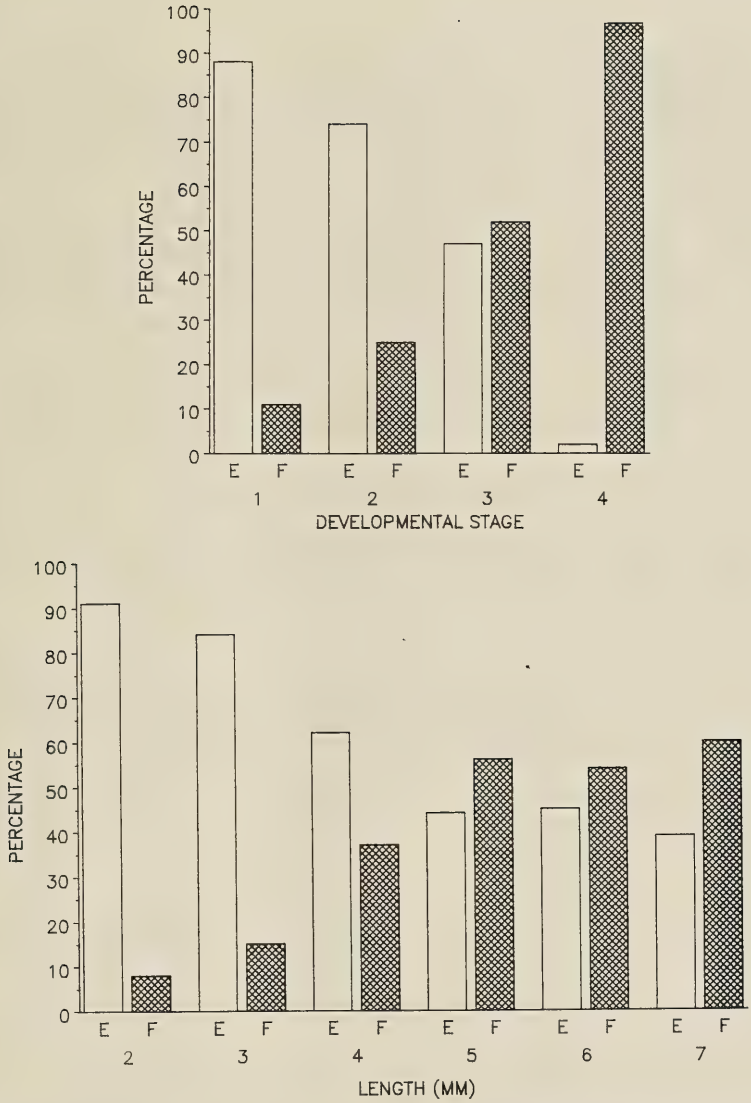


Figure 23. Percent occurrence of each developmental stage and 1-mm size class collected at the mouth of the Niantic River during ebb (E) and flood (F) tidal stages in 1984 and 1985.

The eight tidal cycles sampled since 1983 clearly indicated a net loss of smaller larvae that lacked fin rays and had little or no locomotion. However, larger larvae with developed fin rays apparently utilized vertical migration in relation to tidal currents for passive movement back into the Niantic River. This vertical migration of larvae after fin ray development was also apparent in the 24-h studies conducted at station C in the river (Fig. 12). Other researchers also reported vertical migration in early life history stages of fish. Diel movements of larval yellowtail flounder (*Limanda ferruginea*) were found to increase with larval size (Smith et al. 1978). Atlantic herring larvae synchronized vertical migration with flood tides to minimize seaward transport (Fortier and Leggett 1983). Post-larval spot (*Leiostomus xanthurus*), Atlantic croaker (*Micropogonias undulatus*), and *Paralichthys* spp. flounders used vertical migration in response to tides as a retention mechanism (Weinstein et al. 1980). Larval North Sea plaice (*Pleuronectes platessa*) demonstrated selective horizontal transport by swimming up from the bottom during flood tides and remaining near the bottom during ebb tides (Rijnsdorp et al. 1985). Most winter flounder larvae found in Niantic Bay probably were tidally flushed from the Niantic River during early developmental stages. After fin ray development, at least some of the older larvae in the bay utilized vertical migration in relation to tidal flow to reenter the river and those within the river demonstrated a similar behavior to remain there.

Table 27. Estimated percent return of larval winter flounder on a flood tide that were flushed from the river on an ebb tide presented by size-class with R^2 values of the harmonic regression models.

Size class	Percent return	R^2 of model
3	23.4	0.80
4	60.0	0.66
5	131.8	0.90
6	114.2	0.89
7	140.9	0.88

Post-larval stage

Abundance (age 0)

Post-larval young-of-the-year winter flounder were collected using a 1-m beam trawl from late May through September of 1983-85 at stations LR and CO or WA in the Niantic River. By design, nets of four increasing mesh sizes (0.8 to 6.4 mm) and increasing tow lengths (50 to 100 m) were used to maximize efficiency in catching young as they grew in size and declined in number. The lack of a tickler chain in the beginning of the study in 1983 was subsequently found to have affected catches and resulted in underestimates of abundance for the first 5 wk (NUSCo 1984). Data from that period were not included in the following calculations of abundance or mortality.

Abundance of young winter flounder peaked in mid-June, most likely when larval recruitment began to be offset by mortality (Fig. 24). Catches tended to stabilize by July and appeared to fluctuate about mean levels for the remainder of the season. Although densities for the first month were not known with certainty, young at LR were probably initially more numerous in 1983 than in 1984 or 1985, based on abundance later in the year. This follows the pattern of Stage 4 larval abundance given above (Fig. 14). More variability was also evident for 1983 as only three replicate tows were taken per sampling trip rather than the four made during 1984 and 1985. Densities at CO in 1983 were initially quite high, but quickly fell to levels similar to LR by late June. Sampling at CO was hampered by the buildup of dense mats of *Enteromorpha clathrata*, a filamentous alga. This station was dropped in favor of WA in mid-1984. Catch at WA was also more variable than LR, but densities appeared to be higher there than in the lower river.

Growth and mortality (age 0)

Growth of young was illustrated by changes in weekly mean length (Fig. 25). Less variability was seen in growth than abundance, especially at LR, with relatively small 95% confidence intervals found. Most variation occurred at CO in 1983 with a large group of smaller individuals clustered about one mode joined each week by a few larger specimens. Growth was significantly greater at LR than upriver at CO or WA after mid-June. After a relatively rapid increase from about 12 to 50 mm from May through July, further growth occurred at a slower rate throughout the remainder of summer with little or no increase in weekly means during September.

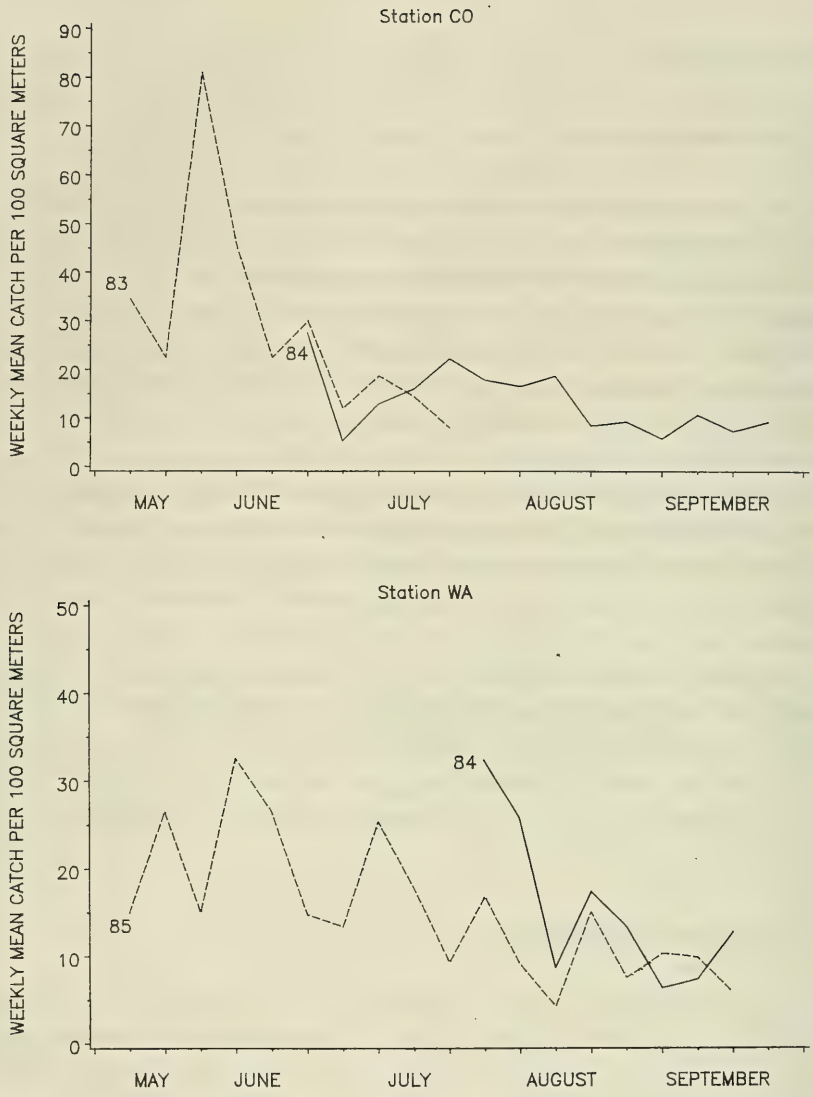


Figure 24. Weekly CPUE of young winter flounder taken in the Niantic River from 1983 through 1985.

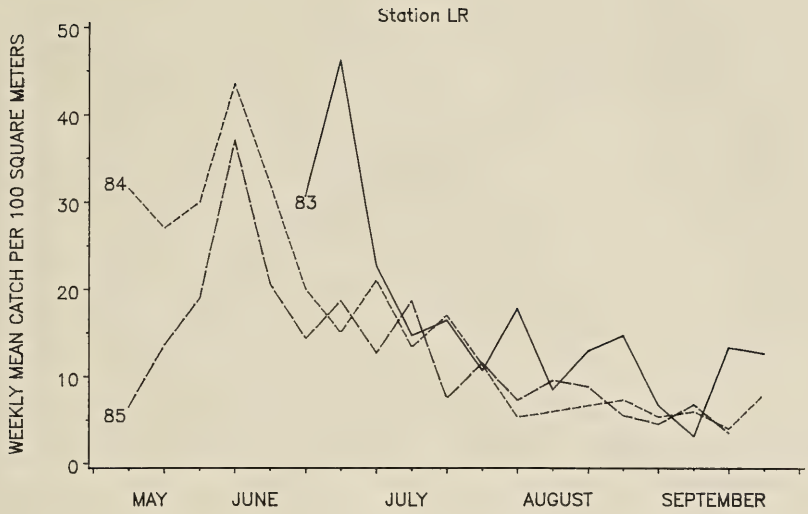


Figure 24. cont.

Weekly means at LR during 1983-85 were similar by year until about late June; means were 6 to 8 mm greater thereafter during 1983 (Fig. 26). Those during 1984 and 1985 were more alike with growth slightly less during the latter year. The reasons for these differences among years are not known. Although water temperatures appeared to be comparable, life history data such as food preference, rates of feeding, and predation upon young are not available. Apparent annual changes in growth may also be caused by differential movement of larger young away from the station, which would also increase the apparent mortality rate. However, only a few average-sized young were taken during the summer at trawl station NR and none at the Niantic Bay stations. Thus, neither large-scale movements nor differential movements by size seemed to have occurred, at least into areas sampled by trawl.

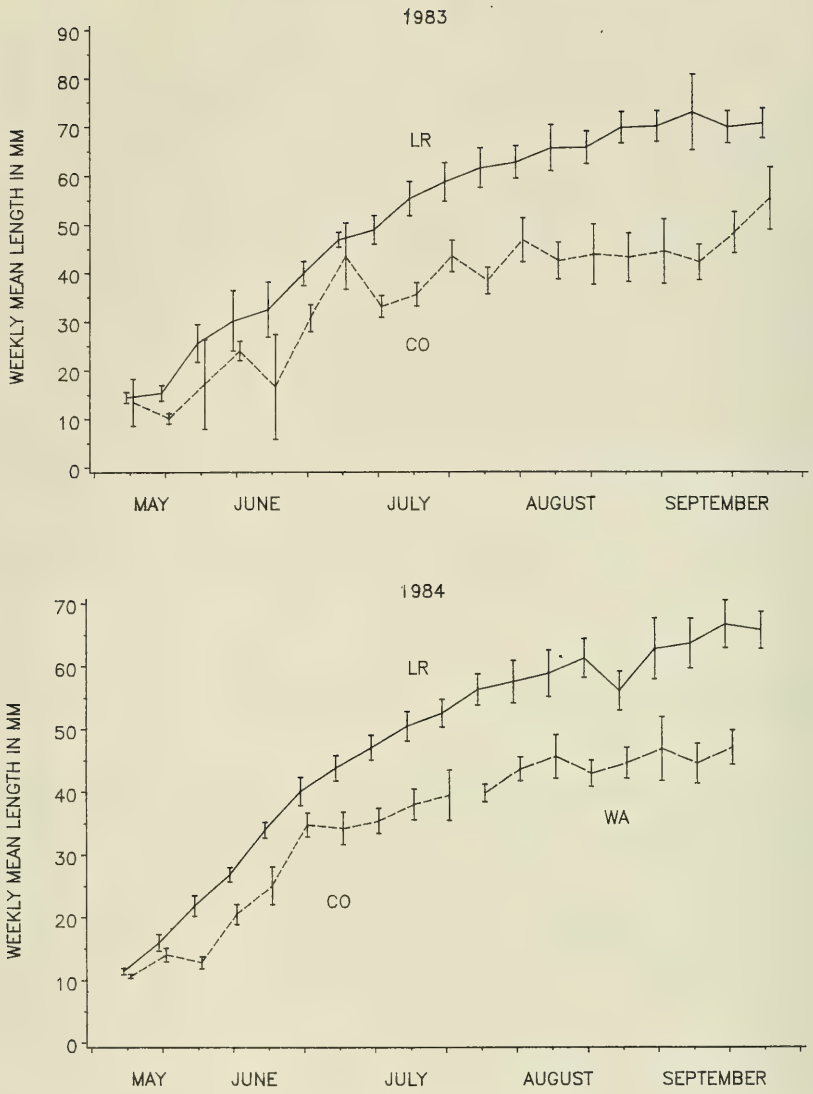


Figure 25. Weekly mean length (±2 standard errors) of young winter flounder taken in the Niantic River from 1983 through 1985.

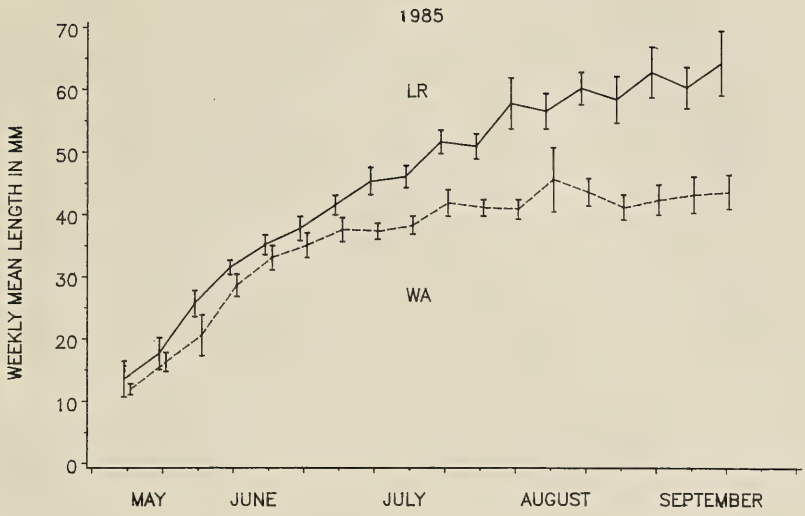


Figure 25. cont.

The instantaneous mortality rate (Z) of post-larval winter flounder was previously reported (NUSCo 1984, 1985, 1986a) using a method described by Jones (1981). Parameters required for this procedure included the cumulative catch of young by length increment and the parameter K of the previously described von Bertalanffy growth model. For young, L_{∞} of the model was fixed at 95 mm for LR and 75 mm for WA fish, or slightly larger than the length of the largest specimen found at each of the stations (NUSCo 1986a). Although not included in this report, analysis of data collected in 1986 indicated that selection of a fixed L_{∞} and the manner in which the cumulative length increments were chosen most likely biased estimates of Z. Therefore, catch curves were constructed using annual abundance data from LR for 1983-85 and WA for 1985. No analysis was attempted for CO due to sampling problems affecting abundance estimates. The catch curve method may also be criticized as young were assumed to have comprised a single-age cohort which was followed from week to week during the sampling season.

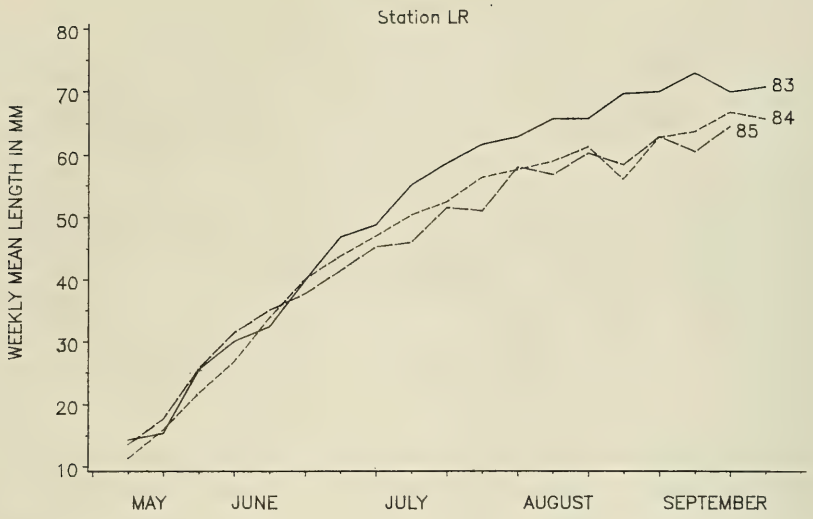


Figure 26. Comparison of weekly mean length of young winter flounder taken at station LR in the Niantic River from 1983 through 1985.

The catch curves for LR had relatively good fits with r^2 ranging from 0.60 to 0.83 (Fig. 27). Remarkably similar values of Z were obtained, resulting in monthly survival estimates of 0.552 to 0.569. A larger estimate of survival was found for WA (0.661). This is in contrast to the more variable survival estimates determined by the method of Jones (1981) and reported in NUSCo (1986a). The monthly survival estimates at station LR in the Niantic River were less than the value of 0.69 reported by Pearcy (1962) for the Mystic River estuary, which is the only published estimate for young winter flounder. Factors affecting the survival of young are unknown, but predation and disease probably cause most deaths. The piscivorous summer flounder (*Paralichthys dentatus*) reached peak abundance during the past decade in the Niantic River during 1984, when it was 2.75 times more numerous than in 1983. However, in 1985 its numbers fell to levels slightly below those in 1983. The abundance of other predators of juvenile winter flounder commonly found during the summer in the Niantic River, such as the double-crested cormorant (*Phalocrocorax auritus*), grubby (*Myoxocephalus aenaeus*), and bluefish *(Pomatomus saltatrix)*, have not been studied.

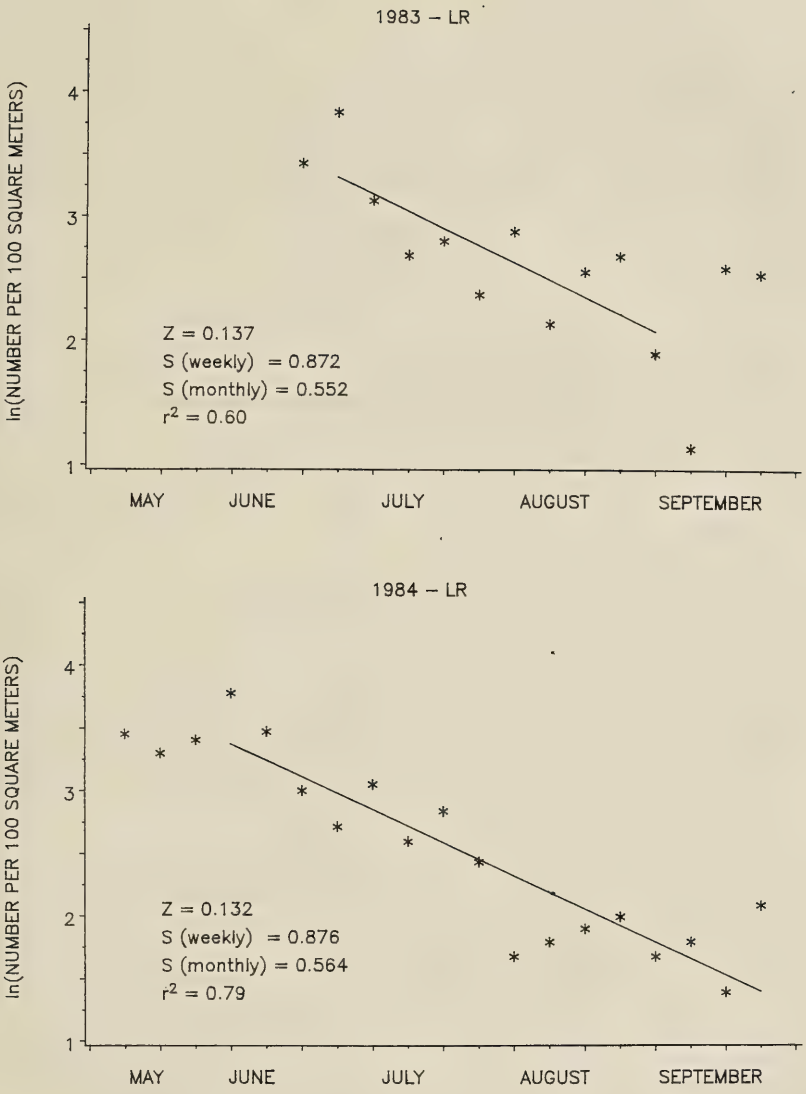
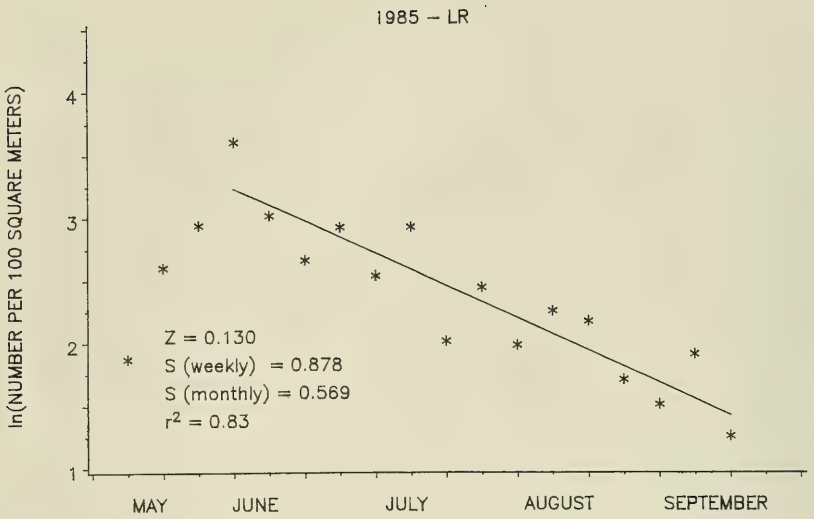


Figure 27. Mortality determined by catch curve for Niantic River young winter flounder from 1983 through 1985.



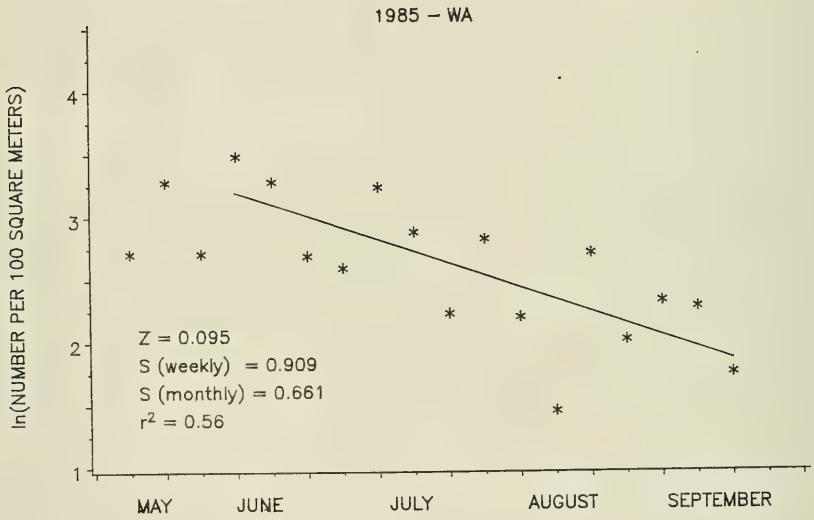


Figure 27. cont.

The microsporidian parasite, *Glugea stephani*, was observed in some individuals collected. Mortality caused by this parasite can be relatively high in winter flounder (Takvorian and Cali 1984; Cali et al. 1986). Incidences of the pathenogenic bacterium *Vibrio anguillarum*, which also can result in fatal infections in winter flounder characterized by fin erosion, dermal ulceration, and hemorrhaging (Levin et al. 1972; Watkins et al. 1981; Sindermann 1985), were not noted.

Abundance (age 1)

An attempt was made in 1981 and 1982 to estimate with the Jolly model the abundance of 6- to 15-cm juvenile winter flounder present in the Niantic River during the spawning season. Fish were marked with freeze-brands during the adult surveys in late winter and released. However, apparent high rates of marking mortality made the Jolly estimates biased and unreliable (NUSCo 1983a) and no further attempts were made to mark small fish.

The median CPUE of juvenile winter flounder smaller than 15 cm in length was calculated for fish taken during the adult winter flounder surveys in the Niantic River from 1976 through 1986 (Table 28, Fig. 28). Nearly all of the fish in this size grouping were age 1 yearlings and represented the year-class spawned during the previous abundance survey. Data were restricted to the mid-March to mid-April period for comparability among years and to stations 1 and 2 because small winter flounder appeared to have been less abundant in the upper river than adults. Inclusion of data from upper river stations could have biased inter-year comparisons because few or no tows were made there prior to 1981.

Juvenile catches were more variable than those of adults and had larger coefficients of variation and skewness. Less uniformity was seen between the median and mean, even after tows were standardized in 1983. Peak abundance was observed in 1981 (1980 year-class), with a median of 87.2. Second and third highest medians were found in following years (61 in 1982, 50.1 in 1983). Abundance declined greatly in 1984 to a median of 16, increased to 27.7 in 1985, and fell to an 11-yr low of 3.6 in 1986. Juvenile abundance, which began to increase in 1979 and peaked in 1981, was generally followed by increasing adult abundance, which peaked in 1982. Abundance of both groups declined through 1984. Juvenile catches were recently reported to have been correlated with adult abundance a year later (NUSCo 1986a), but the addition of 1986 data made this relationship non-significant. Although juvenile CPUE in 1985 was 73% larger than that in 1984, adult abundance did not increase in 1986.

Table 28. Mean and median CPUE of Niantic River winter flounder smaller than 15 cm from 1976 through 1986 during the period of mid-March through mid-April (stations 1 and 2 only).

	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986
Total tows made	80	143	100	79	101	47	39	44	41	48	37
Tows used for CPUE	64	116	84	71	90	45	39	44	41	48	35
% of tows used	80%	81%	84%	90%	90%	96%	100%	100%	100%	100%	95%
Mean CPUE	22.2	31.2	26.4	50.2	55.9	97.7	72.2	50.8	22.9	36.4	6.7
Standard deviation	17.5	26.6	28.5	54.1	43.7	65.2	58.9	35.0	27.2	25.6	7.3
Coeff. of variation	79%	85%	108%	108%	78%	67%	82%	69%	119%	70%	109%
Median CPUE	18	25.5	16.1	27	48.7	87.2	61	50.1	16	27.7	3.6
95% CI	13.5 -25	18.0 -30.9	10.2 -25	17.4 -42.3	33.8 -60	61.0 -120.9	46.5 -86.3	32.8 -61.2	9.9 -20.4	20.9 -41.1	2.5 -8.7
Coeff. of skewness[a]	0.81	1.10	1.81	1.88	1.14	0.67	1.00	0.58	2.71	1.00	1.51

[a] Zero when data are distributed symmetrically

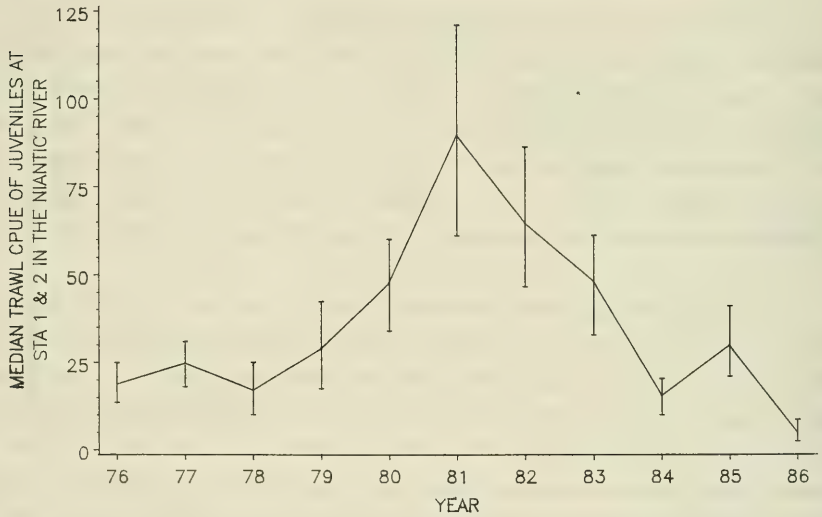


Figure 28. Annual median trawl CPUE (± 2 standard errors) of age 1 winter flounder at stations 1 and 2 in the Niantic River from 1976 through 1986. CPUE based on a tow standardized to a 15-min duration.

The observed abundance of juvenile winter flounder during the 4-wk period may not accurately reflect their absolute abundance. Unlike adults, juveniles do not necessarily enter the river during the spawning season and other factors may influence their movements. However, similar to adults, annual differences in late winter-early spring water temperature did not seemingly correspond with abundance inside or outside of the river. Additional data were examined to determine how accurate the juvenile CPUE was as a predictor of year-class strength. Since many tows were made in the upper river after 1980, median CPUE were calculated which included all tows in the river during the comparable 4-wk time period. These data showed less difference in abundance between 1981-83 and 1984-86 than median CPUE for the lower river channel stations 1 and 2 alone (Fig. 29). This indicated a greater use of the entire river in recent

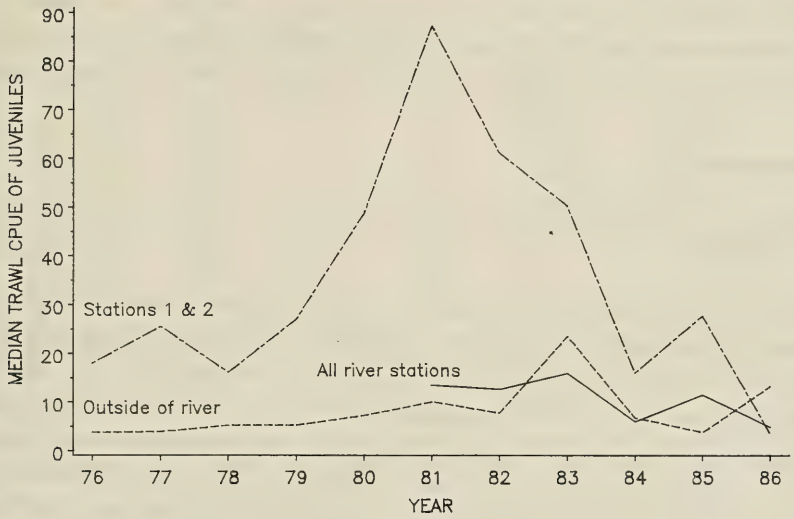


Figure 29. Comparison of median trawl CPUE of age 1 winter flounder at various stations in and outside of the Niantic River from 1976 through 1986. CPUE are consistent within station groupings but are not equivalent (see Materials and Methods).

years by juveniles. As areal distribution increased, concentrations in the lower river most likely decreased. This confounds the use of an abundance index based on tows from only the lower river channel stations.

Juvenile abundance data were also available from the trawl monitoring program for January through April at the five stations outside the river. These CPUE values show generally uniform densities of juveniles from 1976 through 1982 (Fig. 29). A sharp increase occurred in 1983 followed by a decline to average levels in 1984 and 1985 and an increase to another peak in 1986. CPUE values were close in magnitude to those determined for slightly shorter tows in all areas of the river during 1981-86. However, the 1986 median was significantly greater than the values obtained for the lower river channel and for the entire river. The low abundance in the river during 1986 may not be indicative of the true abundance of the 1985 year-class, which may have been less concentrated in the river than others observed. Because of the disparities among the various estimates, the relative abundance of juvenile winter flounder is known with less certainty than that of adults, which are found mainly in the river during winter and early spring.

Impingement

Abundance

Estimates of the number of winter flounder impinged on the traveling screens of MNPS are available from 1972-73 through the present. With the installation of a fish return sluiceway at Unit 1 in December 1983, the the estimated total of 2,926 in 1984-85 was the second lowest of the 13-yr series (Table 29). Most annual estimates for two-unit impingement ranged from 4 to 10 thousand, with an exceptionally high estimate of 24,494 in 1978-79. Since 1976, it has been the second most abundant fish impinged at MNPS, making up 8.4% of the total. If one large impingement event (over 400,000 on one day in July 1984) of sand lance (*Ammodytes americanus*) is ignored, the winter flounder would rank first among all fishes. About two thirds were impinged during winter and relatively few were taken in summer.

Precision of winter flounder impingement estimates should have increased following 1982-83. Using a resource allocation analysis suggested by El-Shamy (1979), the number of samples was changed from a uniform number per month to a variable schedule, specifically reflecting the variability of winter flounder impingement (NUSCo 1983b). More samples were taken in February and March and fewer in other

months, resulting in an approximate 50% reduction in sampling effort (NUSCo 1984).

Table 29. Estimated total number of winter flounder impinged on the intake screens of MNPS by season and year from October 1982 through September 1985 [a]

Year [b]	Fall	Winter	Spring	Summer	Total	% by year	% of all fish
1972-73	405	4,404	1,184	170	6,163	6.2	37.9
1973-74	462	2,663	564	124	3,813	3.8	28.9
1974-75	384	1,666	354	185	2,589	2.6	27.0
1975-76	757	2,409	764	107	4,037	4.0	15.6
1976-77	2,377	4,858	2,250	143	9,628	9.6	32.8
1977-78	371	4,127	1,685	154	6,337	6.3	17.4
1978-79	1,710	17,753	3,884	1,147	24,494	24.5	32.7
1979-80	760	3,377	2,471	309	6,917	6.9	20.0
1980-81	1,050	4,406	1,329	377	7,162	7.2	12.3
1981-82	1,528	6,240	1,529	528	9,825	9.8	15.8
1982-83	578	7,678	2,005	699	10,960	10.9	7.5
1983-84	3,085	1,811	305	106	5,307	5.3	1.0
1984-85	320	2,170	304	132	2,926	2.9	13.2
Total	13,787	63,562	18,628	4,181	100,158	---	---
% by Season	13.8	63.5	18.6	4.2	---	---	---

[a] Estimates for Units 1 and 2 from October 1972 through December 1983 and for Unit 2 thereafter.

[b] October through September

Impingement estimates in themselves do not reflect absolute abundance of a species, but are related to plant design and operational characteristics; time of day; and environmental variables such as water temperature, wave height, wind direction and velocity, and precipitation (Grimes 1975; Landry and Strawn 1975; Lifton and Storr 1978). Severe windstorms combined with falling water temperatures particularly have been correlated with increased impingement of fish at MNPS and elsewhere (Thomas and Miller 1977; Lifton and Storr 1978; NUSCo 1981a, 1983a, 1986a). Eleven instances were found when weekly winter flounder impingement estimates exceeded 1,000 individuals; these samples were examined along with available relevant meteorological data (Table 30). The impingement on these days made up a considerable portion of seasonal and annual estimates. Six of the events occurred in 1979 and accounted

Table 30. Instances of weekly winter flounder impingement estimates of greater than 1,000 individuals and associated meteorological data from October 1972 through September 1985.

	Date	Weekly imp. est.	Mean daily wind speed[a]	Avg. wind direction	Mean daily water temp. (C)	Remarks
1	12-21-76	1,376	21	WNW	4.7	Water temperature declined
	12-22-76		18	W	3.5	2°C during storm
2	3-24-77	1,292	18	NW	2.9	
	3-25-77		18	NW	2.6	
3	1-09-78	1,292	28	SW	4.9	Among the highest average daily wind
	1-10-78		31	W	2.7	speeds since 1975; rapid drop in temperature.
4	1-24-79	2,472	17	ENE	4.3	
	1-25-79		14	S	4.1	
5	1-30-79	1,229	16	NW	4.1	
	1-31-79		12	NW	4.1	
6	2-5-79	7,889	23	WNW	2.3	Water temperature declined 3°C in 3
	2-6-79		21	NW	1.2	days.
7	2-26-79	1,050	12	NE	1.5	
	2-27-79		7	SSW	1.6	
8	3-13-79	2,054	11	S	2.5	
	3-14-79		19	SW	3.0	
	3-15-79		17	WNW	2.5	
9	4-2-79	1,152	20	E	5.0	
10	1-4-82	2,643	24	SE	4.8	
	1-5-82		22	W	4.7	
11	12-28-83	1,559	23	S	5.4	
	12-29-83		14	WNW	5.6	

[a] Miles per hour.

for almost two-thirds of the abnormally high total for that year; one-third of the total alone was from one storm in early February. In most cases the large impingement estimates were associated with sustained daily wind velocities of about 16 mi/h (26 km/h) or more and water temperatures of 5.5 °C or less. Events

associated with winds from the northwest probably resulted from the passage of cold fronts with accompanying falling temperatures and winds from the southwest were perpendicular to the MNPS intakes. The combination of low water temperatures and frequent high winds in winter (NUSCo 1983c), along with less sustained swimming speed and endurance for the winter flounder at colder temperatures (Beamish 1966; Terpin et al. 1977) most likely makes it more susceptible to impingement then. Strong winds alone do not necessarily increase impingement as 3 weeks prior to the December 1983 incident sustained winds of 21 to 35 mi/h from the south to west over a 2-d period only produced a weekly estimate of 145 winter flounder. The water temperature then was 9.5 to 9.8 °C.

Length and sex distribution

The length-frequency distribution of impinged winter flounder by 5-cm length groups showed that the proportion of adults (fish larger than 25 cm) has remained relatively constant and made up about a third of each annual total (Table 31). Catch of mid-sized (15- and 20-cm length groups) and small fish (5 and 10 cm) varied from year to year. Mid-sized fish made up more than half the total in 1977-78, but only one-fifth in 1984-85. About 40% of the catch in 1978-79, 1982-83, and 1984-85 was comprised by small specimens. As noted above, factors such as plant operating conditions and weather influence impingement to a great extent and relative differences in abundance and size-classes impinged each year probably do not reflect actual changes in the winter flounder population.

The sex and reproductive condition were examined for 755 winter flounder impinged from February through April of 1982 and 1,675 from December 1982 through April 1983. The sex ratio in 1982 was 1:1, but males predominated earlier in the season and females later. In 1982-83, 65% were males and 35% females, a ratio opposite to that seen for the Niantic River spawning population. Of the females 25 cm or larger examined from mid-February through April of 1982 and 1983, 56 and 59%, respectively, were gravid. This was contrary to findings from the trawl monitoring program for the same period, which showed relatively few ripe winter flounder outside of the Niantic River. In addition, a comparison of the weekly percentages of gravid females inside the Niantic River with those impinged showed that a much larger portion of the latter had not spawned (Fig. 30). Although sample size was considerably smaller, relatively higher percentages of gravid females continued to be taken at MNPS until the end of April, whereas most fish in the river spawned earlier in the season.

Table 31. Annual mean length and percent length-frequency distribution by 5-cm size intervals of winter flounder impinged at MNPS from October 1976 through September 1985.

Year	Number measured	Mean length (cm)	CV	% length-frequency 5	10	15	20	25	30+
1976-77	4,594	19.5	45%	7	22	16	16	16	23
1977-78	2,653	16.9	43%	12	16	29	23	12	8
1978-79	4,639	16.1	51%	17	23	24	12	13	11
1979-80	2,654	22.3	36%	1	9	25	16	18	31
1980-81	2,197	20.1	45%	7	19	18	13	18	25
1981-82	2,973	20.1	42%	6	10	28	20	13	21
1982-83	3,636	18.0	51%	7	32	19	9	14	20
1983-84	1,213	20.2	41%	5	15	21	21	19	20
1984-85	577	17.8	55%	16	27	10	11	14	23

The occurence of many fish in spawning condition at MNPS may have been partly related to behavior. The environmental cues used by winter flounder to successfully return to the Niantic River from distant areas for spawning are not known. Beverton and Holt (1957) and McKeown (1984) noted that currents are among the important factors in guiding oriented migration of demersal marine fishes. If adult winter flounder move along the shoreline in search of a particular estuary, then the intake currents at MNPS may attract individuals seeking to enter the river for spawning. Also, the larger numbers of males impinged than females may be related to their generally smaller size and therefore lower sustained swimming speeds (Beamish 1966; Terpin et al. 1977), which would have allowed fewer of them to escape from the intake area.

Fish return sluiceways

A fish return sluiceway was installed at MNPS Unit 1 and put into operation in mid-December of 1983. A sluiceway was constructed at Unit 3 and has been in operation since the start of commercial operations in late April of 1986. Survival studies completed before (NUSCo 1981b) and after (NUSCo 1986b) installation of the Unit 1 sluiceway indicated that future impingement mortality of winter flounder at MNPS would most likely be less than 20% (Table 32). Data restricted to only colder months, when

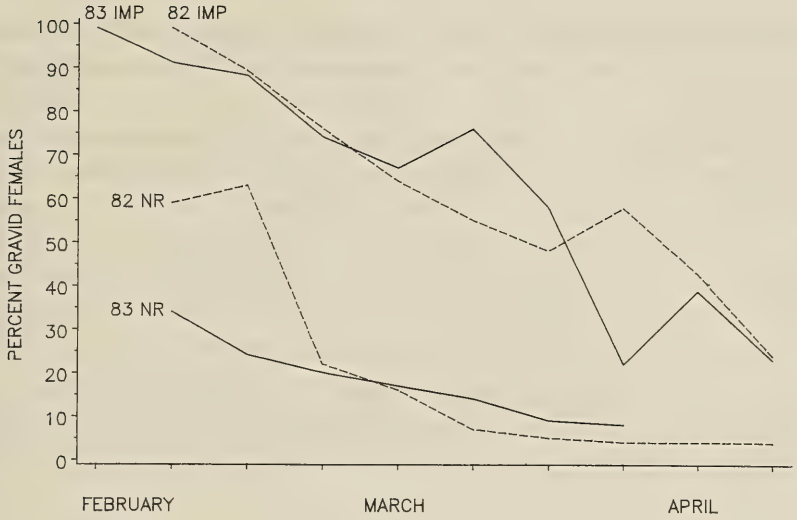


Figure 30. Percentage of adult female winter flounder in spawning condition by week taken in the Niantic River (NR) and on the traveling screens at MNPS (IMP) during 1982 and 1983.

Table 32. Percent initial and extended survival of winter flounder impinged on the MNPS Units 1 and 2 traveling screens.

Date	Time between screenwashes	Number examined	% initial survival	% extended survival[a]	% overall survival
1980-81	Continuous	133	97	100	97
	2 h	146	98	94	92
	4 h	175	97	91	88
	8 h	164	96	95	92
1984-85	6 h	44	93	88	82

[a] 60-h holding period in 1980-81 and 72-h in 1984-85.

most winter flounder were impinged, showed even greater (90%) survival. These estimates were similar to those reported for winter flounder at several other power plants (Tatham et al. 1977; MRI 1982).

One potential problem in returning fish is recirculation and re-impingement, which would increase the probability of mortality. On seven occasions during 1981-83, a total of 299 disc-tagged winter flounder was released near a rock outcrop between Units 1 and 2 (Table 33). Only 15 (5%) fish were impinged, 80% of them within 3 d of release. In comparison, 17 fish were later recaptured by the sport and commercial fisheries (4 to 735 d after release). Only once was more than two fish impinged. This occurred after the June 3, 1981 release, when seven of the nine recaptures for this group were made within 2 d; these were the only fish released at night. A small percentage of winter flounder may be susceptible to re-impingement, especially at night or perhaps during storms. However, the Unit 1 sluiceway terminus is farther from the intakes than the centrally located release point. On ebb tides the flow could carry fish away from the MNPS intakes, depending upon their orientation to currents.

Table 33. Summary of winter flounder tagged and released near the intakes of MNPS Units 1 and 2.

Date of release	Source of fish	Number released	Number recaptured: Impinged	Sport fishery	Commercial fishery	NUSCo trawling[a]	Percent impinged	Percent recaptured
2 June 1981	Trawling	109	1	7	1	7	0.9	14.7
3 June 1981	Trawling	89	9	1	1	0	10.1	12.4
29 Mar. 1982	Impingement	15	0	3	1	1	0	33.3
5 Apr. 1982	Impingement	18	1	0	0	0	5.6	5.6
12 Apr. 1982	Impingement	8	1	0	0	0	12.5	12.5
26 Apr. 1982	Impingement	20	1	2	1	0	5.0	20.0
7 Mar. 1983	Impingement	40	2	0	0	2	5.0	10.0
Total		299	15	13	4	10	5.0	14.0

[a] All released alive; 3 subsequently recaptured a second time and included in above totals.

The sluiceways at Units 1 and 3 should substantially reduce the impact of impingement on juvenile and adult winter flounder. This is particularly important as apparently a relatively large proportion of impinged adults did not spawn before encountering the intakes. Studies at NUEL are currently underway to examine the effectiveness of the Unit 3 sluiceway in returning winter flounder and other species to Niantic Bay.

Entrainment

Abundance

Sampling at the discharges of MNPS Units 1 and 2 to estimate the number of winter flounder larvae entrained through the condenser cooling water system has been conducted since 1976. This is the longest time-series of data at NUEL with consistent yearly sampling on the abundance of larval winter flounder. Generally, larvae were entrained from late February through the end of June with greatest densities from mid-April through May. Over 60% of the larvae entrained during the 10-yr period were 5.0 mm and larger (Fig. 31). Since 1983, the proportion of Stage 1 larvae entrained was 2%, Stage 2 was 31%, Stage 3 was 56%, and Stage 4 was 11%. As stated previously, smaller larvae are not abundant in the Niantic Bay and therefore are less susceptible to entrainment.

Since 1982, the median has been used as a measure of the annual entrainment density and in calculating the total number of winter flounder larvae entrained. A median was used instead of a mean because the data were highly skewed and the median was a better estimate of central tendency (NUSCo 1983a). Selection of the time period used to calculate the median was changed for this report. Previously, the median calculations were based on an annual time period selected by examination of the annual density distribution over time with an arbitrary determination of the beginning and end of the season. For this report, the season was determined as the period in which 95% of the total cumulative abundance occurred, excluding samples containing the first and last 2.5% of the cumulative abundance over time. This method reduced the number of zero density values used in the calculations. Prior to 1982, entrainment estimates were based on weekly means multiplied by the total weekly volume of water passing through MNPS. These weekly values were then summed for an annual estimate, but confidence intervals could not be calculated. Comparison of the results of these three methods (NUSCo 1983c, 1986a, and this report) indicated large discrepancies among the the estimates. Further evaluations of techniques for estimating

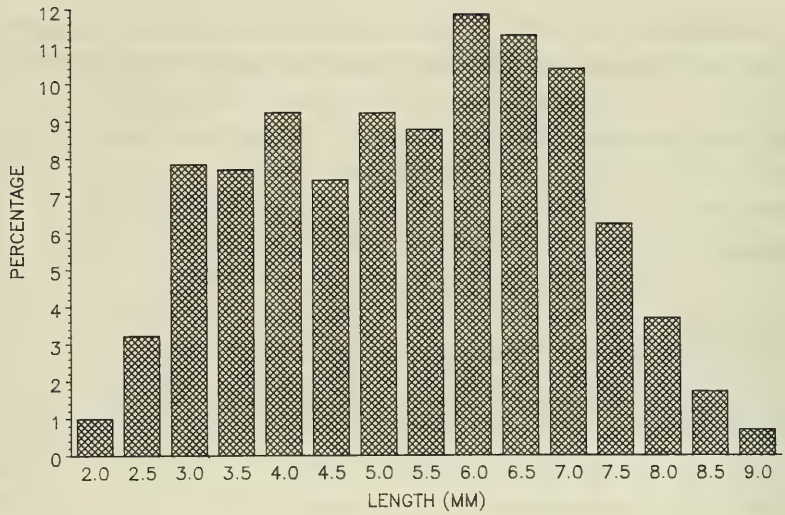


Figure 31. Length-frequency distribution of larval winter flounder entrained at MNPS from 1976 through 1985.

annual entrainment numbers should be undertaken to establish the most credible estimates of entrainment for impact assessment.

The median entrainment density has declined since 1983, which agreed with the decline in the cumulative frequency in Stage 2 through 4 larvae at this station (Table 34; Fig. 14). The median densities may be grouped into high and low years based on the lack of overlap of the 95% confidence intervals with three low years (1977, 1978, and 1979) and four high years (1976, 1980, 1982, and 1983). The confidence intervals of the remaining three years (1981, 1984, and 1985) overlapped either or both of the low or high groups. A similar but slightly different grouping of actual entrainment estimates was found with three low years (1977, 1979, and 1981) and four high years (1976, 1980, 1982, and 1983), with remaining years overlapping. The total entrainment estimates took into account the volume of water withdrawn for condenser cooling and may vary according to plant operations.

Table 34. Yearly median densities (n per 500 m^3) of winter flounder larvae in entrainment samples during their season of occurrence and annual entrainment estimates with 95% confidence intervals for MNPS Units 1 and 2 from 1976 through 1985.

Year	Median	95% CI	Estimate ($x10^6$)	95% CI ($x10^6$)
1976	158.0	114.2 - 185.1	95.5	69.0 - 111.9
1977	68.3	54.5 - 96.3	30.9	24.7 - 43.6
1978	86.6	65.0 - 106.4	58.4	43.8 - 71.7
1979	90.3	69.7 - 108.4	36.6	28.2 - 43.9
1980	201.5	163.6 - 234.6	140.1	113.7 - 163.1
1981	139.2	99.3 - 182.6	47.6	33.9 - 62.4
1982	183.5	147.5 - 215.1	137.1	110.3 - 160.7
1983	244.4	158.1 - 314.8	172.9	111.9 - 222.6
1984	185.5	107.5 - 226.2	90.0	52.2 - 109.8
1985	107.1	78.8 - 149.5	65.9	48.5 - 92.1

The 10 yr of entrainment data provided an opportunity to examine factors that would affect annual abundance and the timing of peak entrainment. The Gompertz function (Equation 6) was fit to the data to compare the α parameter as an index of abundance to the median entrainment density and to compare annual estimated dates of peak abundance (Table 35). A high correlation was found between the median and the α parameter (Spearman's rank correlation coefficient = 0.855; $p = 0.0016$), showing that the α parameter was a good index of abundance. The annual abundance of entrained larvae appeared to be related to annual egg production in the river (Fig. 32). A linear regression was fitted to the total egg production index (Table 11) versus the α parameter. The 1980 total egg production index was excluded from the analyses because of the previously discussed problems with data from that year. The relationship between egg production and entrainment abundance suggested that natural mortality from hatching to the time of entrainment was similar for the 8 yr examined.

Table 35. The α parameter as an index of abundance and the date of inflection as an estimate of the date of peak abundance from the Gompertz function for larval winter flounder entrained from 1976 through 1985.

Year	α	Date of inflection
1976	2321	13 Apr
1977	1263	18 May
1978	2476	8 May
1979	1665	1 May
1980	3216	24 Apr
1981	1693	5 May
1982	3012	28 Apr
1983	3565	18 Apr
1984	2568	27 Apr
1985	1951	17 Apr

The estimated date of peak density at EN varied from April 13 in 1976 to May 18 in 1977. For the period of 1981-85, a similar pattern in the timing of peak abundance was also evident in the abundance curves for Niantic Bay with stations NB and EN combined (Fig. 13). The estimated date of peak abundance was determined from the inflection point of Gompertz function. This inflection was caused by the decline in larval abundance and for EN data the decline was partly caused by the decrease of larvae through recruitment to the juvenile stage. Because water temperature could affect the rate of development to juveniles, temperatures during the entrainment season were compared to the timing of peak abundance (Fig. 33). Temperature was expressed as the yearly deviation during March through May from the average temperature for the 10-yr period. Time of peak abundance was the number of days after February 15 that the peak occurred each year. It appeared that as water temperature increased, the date of peak abundance was earlier. This could be related to faster developmental rate to the juvenile stage or changes in annual time of spawning (Fig. 5) due to water temperature. Although other factors could affect these parameters, it was apparent that numbers entrained were probably related to total egg production in the Niantic River and the length of time a larva was susceptible to entrainment, which was in turn related to

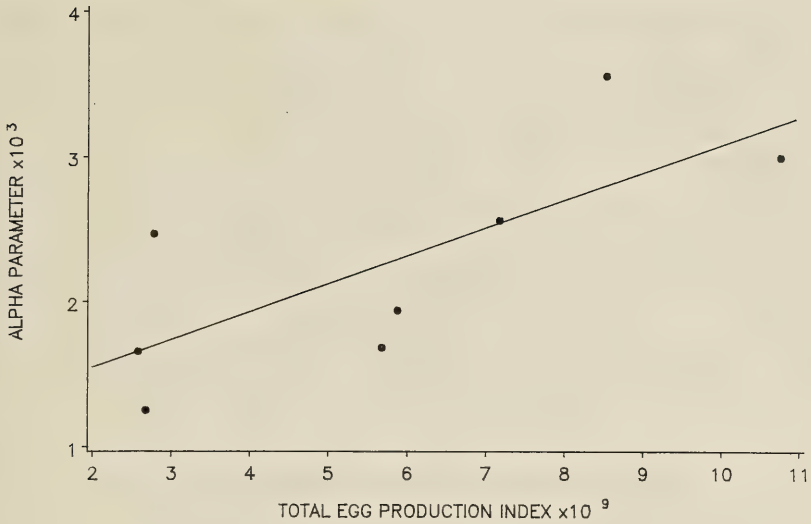


Figure 32. The relationship between the α parameter (y) as an index of the annual abundance of winter flounder larvae entrained and the annual total egg production index (x) in the Niantic River from 1977 through 1985 with the fitted linear regression of: $y = 1164.7 + 191.8\ (x)$; $r^2 = 0.56$.

water temperature affecting growth and development.

Survival

Thermal tolerance studies on larval winter flounder were conducted to estimate the effect of increased temperatures on larvae during entrainment (NUSCo 1975). Larvae were grouped as pre-metamorphosed (less than 5.0 mm) and metamorphosing (5.0 mm and larger). Larvae were exposed to a ΔT of 13 °C using an acclimation temperature of 8 °C. Mortality increased with exposure time for pre-metamorphosed larvae from 29% at 1 h, 48% at 2 h, 53% at 3 h, to 89% at 6 h. No mortality occurred for metamorphosing larvae with exposures up to 9 h. Because most larvae entrained were 5.0 mm and larger (Fig. 31) and

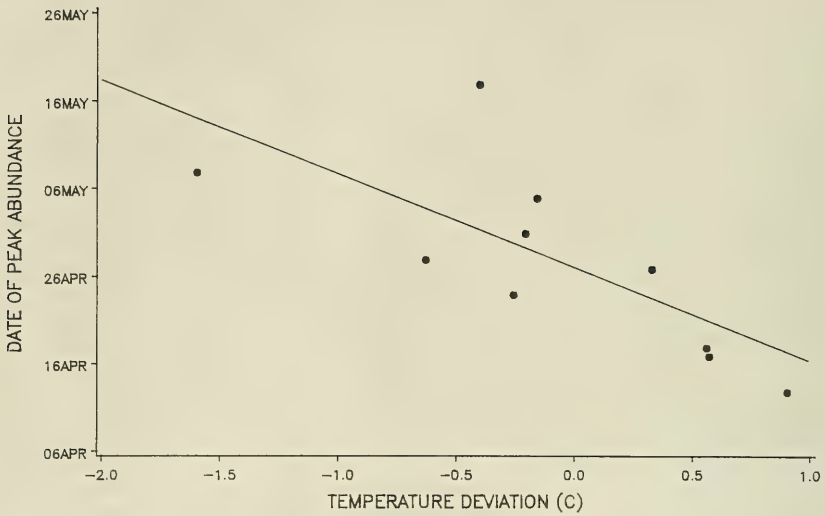


Figure 33. The relationship between the estimated annual date of peak abundance (y) of entrained winter flounder larvae and the annual temperature deviation (x) during March-May with the fitted linear regression of: $y = 72.0 - 10.7\ (x)$; $r^2 = 0.53$.

the retention time in the quarry with 2-unit operations was approximately 3 h, most larvae would survive the thermal increase due to entrainment, but not necessarily any mechanical damage.

The critical thermal maximum was determined by increasing the water temperature from 8 °C at a rate of 1 °C per min until total mortality occurred. The critical thermal maximum was 25 to 26 °C for pre-metamorphosed and 24 to 26 °C for metamorphosing larvae. Water temperatures in the quarry generally reached or exceeded 24 °C by mid-May. Whether the critical thermal maximum temperature actually represents the maximum temperature that larval winter flounder can survive is doubtful because an almost 15 °C rise in less than 20 min would not provide a period for acclimation at each temperature. Furthermore, the CTM may vary according to any particular acclimation temperature (NUSCo 1975).

Entrainment mortality studies were conducted in 1983 on larvae that had passed through the plant (NUSCo 1984). A total of 135 winter flounder larvae was collected during 11 sampling sessions (Table 36). During the study the effluent ΔT ranged from 8.0 to 11.5°C. Of the 24 Stage 2 larvae collected, 8 were alive following capture but none survived the 2- to 4-h effluent holding period. Stage 3 larvae had greater survival than Stage 2 following capture and the effluent holding period, but all survivors died during the first 24 h of the latent holding period. All 24 Stage 4 larvae were alive following capture, with 19 (79%) surviving effluent and 96-h latent holding periods. From 1983 though 1985, 11% of the larvae entrained were Stage 4 and many of these most likely would have survived passage. These older larvae would also have had a greater probability of recruitment to the adult stock than the younger stages. Survival of larger winter flounder larvae entrained would reduce the estimated effects in previous assessments (Hess et al. 1975; Saila 1976; NUSCo 1983c), which assumed 100% mortality for entrained larvae.

Table 36. Results of larval winter flounder entrainment mortality studies.

Stage	Number captured	Alive at capture	Surviving effluent holding	Surviving 96-h latent holding
2	24	8	0	0
3	87	69	15	0
4	24	24	19	19
Total	135	101	34	19

Impact assessment

The potential impact of the 3-unit MNPS over its operational lifetime on the Niantic River population of winter flounder has been addressed by a deterministic impact assessment model. This model was developed under the direction of Dr. Saul Saila of the University of Rhode Island and has been described in numerous reports to NUSCo as well as in the scientific literature (Sissenwine et al. 1973, 1974, 1975; Hess et al. 1975; Vaughan et al. 1976; Saila 1976). The model is subdivided into hydrodynamic, concentration, and population submodels and examines the entrainment of winter flounder larvae, which are

assumed to come from the Niantic River, as the major impact of MNPS operations. The latest application of the model was for the MNPS Unit 3 Environmental Report - Operating License Stage (NUSCo 1983c). Based on data from the literature as well as from NUEL studies at the time of the model run, a potential 5 to 6% decrease in winter flounder abundance was projected to occur after 35 yr of plant operations. The population would recover to within 1% of the equilibrium level after an additional 65-yr period.

It was emphasized in NUSCo (1983c) that the above model results were probably conservative. The effects of entrainment were overestimated since vertical stratification of larvae and vertical variations in current velocity were ignored, as was the potential for a fraction of the entrained larvae to survive, and because of input of larvae from outside the Niantic River. The immigration of winter flounder from other stocks into the area; density-dependent growth, fecundity, adult mortality; and, in some cases, density-dependent larval mortality were not considered. Criticisms could be made of many of the model assumptions, particularly the ones made for estimating the Ricker (1954) stock-recruitment function parameters. Nevertheless, the model has been independently evaluated and found to be an acceptable, conservative method of assessing MNPS impact (Gore et al. 1977; Thomas et al. 1978).

Some mathematical representation of the recruitment process is common to many fisheries population models. Since egg and larval survival is more dependent upon environmental factors than adult survival, a prominant feature of recruitment data for many species is the large amount of variability present (Jones 1982). In spite of this, deterministic population models use recruitment equations which have constant parameters and assume either no adult age-structure or stable age-structure implying equilibrium. However, models that take into account the effects of natural variability in the recruitment process have become available recently. A stochastic population dynamics model, based on work by Lorda (1982) and Reed et al. (1984), is currently being modified at NUEL to incorporate detailed early life history and temperature effects on larval survival. This model explicitly uses the natural variability of some key population parameters and data collected specifically for the Niantic River winter flounder stock. In recent years, increased emphasis has been placed on understanding the dynamics of the early life history stages. The factors governing the production and mortality of larvae should be known before assessing the impact of MNPS operations. In particular, the quantification of larval mortality and its partition into density-dependent and -independent components can be expected to be critical for the realistic estimation of entrainment effects. Model results will include a probabilistic risk assessment analysis, which should provide better

and more realistic estimate of potential losses to the winter flounder population caused by the operation of MNPS.

CONCLUSIONS

The Niantic River population of the winter flounder has been studied since 1976 because of its importance to the sport and commercial fisheries of Connecticut and the potential for impact by MNPS operations, especially from entrainment of larvae. Studies of adult winter flounder spawning in the river have provided a time-series of data for impact assessment, including adult stock size, reproduction, movements, exploitation, and rates of growth and mortality. Similarly, estimates of impingement and entrainment during the same time period were available to measure impact. Since 1983, increased emphasis was placed on understanding critical early life history stages of winter flounder in the Niantic River. To date, there is no evidence that MNPS has significantly affected the winter flounder population. Periodic cycles in abundance are typical for this species throughout its range. Although numbers of adults have declined in recent years, the decrease apparently has been due to natural causes or perhaps from an increase in commercial fishing in Southern New England.

Events during the larval stage within the Niantic River are most important in determining the success of a year-class and their understanding is still incomplete. The factors which govern the production and mortality of larvae must be known before assessing the impact of power plant operations. Quantifying larval mortality and partitioning it into density-dependent and -independent components will allow the assessment of entrainment effects. Similarly, knowledge of the post-larval juvenile stage is also important. The mortality of these fish is intermediate between that of larvae and adults and may play an important part in establishing the success of a year-class. Impingement of juveniles and adults at MNPS has become less of a concern as the installation of fish return sluiceways at Units 1 and 3 has reduced this impact. Recent decreases in impingement were most likely a result of the general decline in abundance, rather than its cause.

MNPS Unit 3 began commercial operations in late April of 1986, immediately following the adult winter flounder spawning season. Therefore, the data gathered up to that time will serve as the baseline for assessing the full impact of 3-unit operations of MNPS. Future work will focus on larval, juvenile,

and adult population parameters. The development of a stochastic population dynamics model for impact assessment will continue to be an important goal for NUEL.

SUMMARY

1. The life history and population dynamics of the winter flounder has been studied intensively since 1973, due to its importance to the sport and commercial fisheries of Connecticut and potential for impact. Because winter flounder stocks are localized, most work has concentrated on the population spawning in the Niantic River to determine if MNPS impacts of impingement and entrainment have caused or would cause changes in abundance beyond those expected from natural variation.

2. Annual estimates of the Niantic River spawning population have been made since 1976. An abundance index based on the Jolly (1965) model showed that numbers were relatively stable from 1976 through 1980, increased to a peak in 1982, and subsequently declined to an 11-yr low in 1986. Abundance determined by trawl CPUE generally paralleled the Jolly index through 1982. The decline in CPUE was greater through 1985 and less in 1986 than for the corresponding Jolly estimates.

3. Evaluation of both abundance estimators indicated that bias due to failures to meet assumptions and errors due to low sampling intensity may have affected the Jolly estimates. CPUE values may have been influenced by changes in sampling methodologies, varying distribution of winter flounder, and variable annual conditions in the Niantic River.

4. Data from the trawl monitoring program were used with time-based harmonic regression models. However, most models were unsatisfactory due to insufficient data or a lack of a repetitive pattern of abundance. High variability in catch, relatively low effort, and the mixture of stocks found at most stations at certain times of the year make these trawl data difficult to interpret and of limited use in assessing MNPS impact.

5. Throughout Southern New England, winter flounder abundance has recently declined because of natural fluctuations and also most likely from increases in commercial fishing.

6. Similar to other populations, the average sex ratio for Niantic River winter flounder was about 1.44 females for each male. The length of 50% maturation of females was 26.8 cm, equivalent to age 3 or 4.

7. Most spawning in the Niantic River was completed by early April with annual variations apparently related to water temperature. Egg production was a function of female size and the length-fecundity relationship was similar to those reported for other populations. Egg production peaked in 1982 and has since decreased about 80%.

8. Scales were successfully used to age winter flounder. Mean lengths of age 3 and older females were significantly larger than those of males. Growth was relatively rapid in early years, but older age groups overlapped considerably in size. Growth of the Niantic River fish was less than other populations in the region through age 2, but equaled or exceeded their means at age 3 and older.

9. The von Bertalanffy model was used to calculate population growth parameters using 1983 length-at-age data. L_∞ was determined as 423 and 381 mm and K as 0.42 and 0.44 for females and males, respectively.

10. The mean annual survival rate of adults age 3 and older was determined as 0.486 using a catch curve with samples combined from successive years to reduce bias.

11. As found elsewhere, the winter flounder preyed upon a variety of benthic organisms and algae. Food items varied by location and reflected bottom type and different benthic communities.

12. The overall rate of return of Petersen disc-tagged winter flounder was 25%. About twice as many were taken by the sport than the commercial fishery, although less cooperation was probably received from the latter. Most (70%) of the returns were from local waters and three times as many of the longer-distance recaptures were made in waters to the east than to the west.

13. Direct tissue isoelectric focusing techniques were used to differentiate stocks of winter flounder. Good separation was achieved using fish from major estuaries in Connecticut and Rhode Island at least 8 km apart. A second study using fish from areas closer to MNPS showed more homogeneity, with

significant intermixing occurring throughout much of the year. The technique could not be used to separate immature specimens.

14. Several special studies and analyses were conducted to identify possible sampling biases in the larval winter flounder data base. The results of these studies included reduced larval net extrusion with 202-µm mesh nets compared to 333- and 505-µm nets, increased sample density of larger larvae in night collections, and changes in sample densities in relation to tidal stage at a station in the lower portion of the Niantic River. Due to the identified sample biases, much of the offshore data collected prior to 1980 could not be used to examine the life history of larval winter flounder.

15. Based on the abundance and distribution of smaller larvae, spawning primarily occurred in the Niantic River. Larvae were gradually flushed into Niantic Bay, where larger larvae dominated. The spatial distribution of larvae within the Niantic River varied from year to year, but generally smaller larvae were more prevalent in the upper portion of the river and larger larvae in the lower. The lion's mane jellyfish was identified as an important predator of larval winter flounder.

16. Examination of otoliths from field-collected and laboratory-reared winter flounder larvae indicted that daily increments were not visible. Based on the length-frequency distribution, most larval mortality occurred at the time of first feeding (3-4 mm). Transition to the dermersal juvenile stage occurred at about 6-7 mm.

17. Eight tidal export-import studies were conducted at the mouth of the Niantic River during 1983-85. The results showed a net export of 4 mm and smaller winter flounder larvae and a net import of 5 mm and larger larvae. Larvae with developed fin rays migrated vertically in response to tidal currents to reenter the Niantic River and those within the river demonstrated a similar behavior as a retention mechanism.

18. Abundance of post-larval young-of-the-year peaked in mid-June and stabilized by late July. Young were most numerous in the lower river during 1983, with similar densities found during 1984 and 1985.

19. Growth of young in the lower river was significantly greater than at stations upriver after mid-June. Weekly mean lengths in 1983 were about 6 to 8 mm larger than in 1984 or 1985. Monthly survival estimates of young ranged from 0.552 to 0.569.

20. Peak abundance of age 1 juvenile winter flounder taken in the Niantic River during the adult surveys occurred in 1981, with second and third highest CPUE in following years. An 11-yr low was found in 1986. However, in recent years juveniles have been found in more areas throughout the river and in Niantic Bay during the time of the surveys. This variation in distribution makes the estimation of juvenile abundance less certain than that of adults.

21. About two-thirds of the total number of winter flounder impinged on the traveling screens of MNPS were taken in winter. Before 1984, annual estimates usually ranged from 4 to 10 thousand with winter storms accounting for large proportions of most annual totals.

22. Sex ratios and reproductive condition of impinged fish differed from fish taken in the river. The predominance of males and of gravid females in the collections indicated that at times impingement was related to behavior of winter flounder.

23. A fish return sluiceway was installed at MNPS Unit 1 in December of 1983 and studies showed that survival of returned winter flounder would be considerable (ca. 80-90%). This greatly reduces the impact of impingement on the winter flounder.

24. Entrainment sampling has been conducted since 1976. A majority (>60%) of the winter flounder larvae entrained were 5 mm and larger. The greatest entrainment densities occurred from mid-April through May. Based on the median annual entrainment density, three years were low (1977-79), four years were high (1976, 1980-83), and the remaining years were intermediate. Annual entrainment was related to total egg production in the Niantic River and the length of time a larva was susceptible to entrainment was related to water temperature.

25. The effects of entrainment on larval winter flounder were examined in the laboratory and field. Larvae 5 mm and larger were able to survive a ΔT of 13° C for up to 9 h. The estimated critical thermal

maxmium was approximately 24° C. A mortality study showed that about 80% of Stage 4 larvae entrained would have survived entrainment.

26. Impact assessment was addressed using a deterministic model developed by the University of Rhode Island. The model, subdivided into hydrodynamic, concentration, and population submodels, predicted a 5 to 6% decrease in the Niantic River population after 35 yr of MNPS operations. Based on the initial assumptions, the model results were probably conservative. A new stochastic population dynamics model, which takes into account the natural variability in the recruitment process, is currently under development at NUEL and will provide a more realistic estimate of potential losses.

27. To date, there is no evidence that MNPS has significantly affected the local winter flounder population. Variability in annual abundance appears to be related to natural events and has been noted throughout the region. Future work at NUEL will focus on the early life history stage, including estimates of larval mortality, which is critical to the understanding and estimation of any deleterious effects of MNPS.

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Appendix I. Weekly catch data used for estimating population abundance of Niantic River winter flounder during 1976.

Week no.	Date-week of	Total catch	No. unmrk'd	No. marked	No. removed	No. exam'd	Recaps 1975	Recaptures (week marked) 1 2 3 4 5 6 7	Total recap
1	3-1	1,352	329	1,021	2	1,023	18	-	-
2	3-8	2,379	690	1,670	19	1,689	16	40 -	40
3	3-15	1,292	283	1,004	5	1,009	42	22 64 -	86
4	3-22	1,617	627	982	8	990	28	15 42 36 -	93
5	3-29	1,283	428	853	2	855	9	6 19 13 19 -	57
6	4-5	1,496	450	949	97	1,046	10	8 16 5 24 33 -	86
7	4-12	1,379	1,372	---	7	1,023	11	7 11 6 10 22 35 -	91
Total		10,798	4,179	6,479	140	7,635	131	98 152 60 53 55 35 -	453

Appendix II. The 1976 abundance estimate of winter flounder larger than 15 cm during the spawning period in the Niantic River.

Date-wk. of	Total number (N)	Standard error of N	Probability of survival (Φ)	Standard error of Φ	Calculated no. joining (B)	Standard error of B
3-1			0.663	0.093		
3-8	28,596	5,870	0.937	0.142	-1,411	5,474
3-15	25,353	4,511	0.578	0.110	4,292	2,535
3-22	18,939	3,524	0.527	0.102	11,116	3,373
3-29	21,094	4,286	0.728	0.164	4,148	3,012
4-5	19,514	4,371				

Abundance index $= \frac{(N_2 + N_3 + N_4)}{3}$ 21,795

± 2 standard errors $= \frac{2}{3}\sqrt{VarN_2 + VarN_3 + VarN_4}$ $\pm 4,768$

Appendix III. Weekly catch data used for determining population abundance of Niantic River winter flounder during 1977.

Week no.	Date-week of	Total catch	No. unmrk'd	No. marked	No. removed	No. exam'd	Recaps 1975-76	Recaptures (week marked) 1	2	3	4	5	6	Total recap
1	3-7	616	91	486	39	525	8	-						-
2	3-14	1,975	925	986	64	1,050	6	9	-					9
3	3-21	2,219	1,242	916	61	977	7	13	35	-				48
4	3-28	1,596	834	762	0	762	4	1	23	22	-			46
5	4-4	971	267	587	117	704	7	2	18	26	35	-		81
6	4-11	1,536	1,535	-	1	907	6	5	9	22	21	16	-	73
Total		8,913	4,894	3,737	282	4,925	38	30	85	70	56	16	-	257

Appendix IV. The 1977 abundance estimate of winter flounder larger than 15 cm during the spawning period in the Niantic River.

Date-wk. of	Total number (N)	Standard error of N	Probability of survival (Φ)	Standard error of Φ	Calculated no. joining (B)	Standard error of B
3-7			0.520	0.119		
3-14	29,470	11,666	0.656	0.110	-2,874	7,554
3-21	16,425	3,462	0.694	0.124	7,895	3,496
3-28	19,245	4,097	1.157	0.331	-3,383	4,015
4-4	18,879	5,429				

Abundance index $= \frac{(N_2 + N_3 + N_4)}{3}$ 18,183

± 2 standard errors $= \frac{2}{3}\sqrt{VarN_2 + VarN_3 + VarN_4}$ $\pm 5,088$

Appendix V. Weekly catch data used for determining population abundance of Niantic River winter flounder during 1978.

Week no.	Date-week of	Total catch	No. unmrk'd	No. marked	No. removed	No. exam'd	Recaps 1975-77	Recaptures (week marked) 1 2 3 4 5 6 7 8	Total recap
1	3-6	108	40	68	0	68	3	-	-
2	3-13	321	96	189	36	225	13	0 -	0
3	3-20	1,083	647	436	0	436	14	0 4 -	4
4	3-27	1,108	576	517	15	532	4	0 0 3 -	3
5	4-3	2,037	992	1,045	0	1,045	15	0 3 20 15 -	38
6	4-10	1,893	842	1,051	0	1,051	14	3 2 8 7 35 -	55
7	4-17	1,973	862	1,111	0	1,111	6	0 6 6 6 34 57 -	109
8	4-24	2,486	2,485	-	1	1,574	13	0 2 1 5 21 33 89 -	151
Total		11,009	6,540	4,417	52	6,042	82	3 17 38 33 90 90 89 -	360

Appendix VI. The 1978 abundance estimate of winter flounder larger than 15 cm during the spawning period in the Niantic River.

Date-wk. of	Total number (N)	Standard error of N	Probability of survival (Φ)	Standard error of Φ	Calculated no. joining (B)	Standard error of B
3-6 to 3-27			0.476	0.078		
4-3	15,733	3,513	0.634	0.096	9,154	3,318
4-10	19,126	3,601	0.442	0.070	543	1,463
4-17	9,000	1,470				

Abundance index $= \frac{(N_1 + N_2 + N_3)}{3}$ 14,620

± 2 standard errors $= \frac{2}{3}\sqrt{VarN_1 + VarN_2 + VarN_3}$ $\pm 3,494$

Appendix VII. Weekly catch data used for determining population abundance of Niantic River winter flounder during 1979.

Week no.	Date-week of	Total catch	No. unmrk'd	No. marked	No. removed	No. exam'd	Recaps 1975-78	Recaptures (week marked)						Total recap
								1	2	3	4	5	6	
1	3-12	384	163	221	0	221	3	-						-
2	3-19	1,170	613	554	3	557	15	1	-					1
3	3-26	2,781	1,735	1,046	0	1,046	6	2	4	-				6
4	4-2	2,983	1,674	1,308	1	1,309	3	2	5	43	-			50
5	4-9	3,317	2,379	938	0	938	4	3	5	27	51	-		86
6	4-16	1,383	1,381	-	2	670	4	1	1	20	26	50	-	98
Total		12,018	7,945	4,067	6	4,741	35	9	15	90	77	50	-	241

Appendix VIII. The 1979 abundance estimate of winter flounder larger than 15 cm during the spawning period in the Niantic River.

Date-wk. of	Total number (N)	Standard error of N	Probability of survival (Φ)	Standard error of Φ	Calculated no. joining (B)	Standard error of B
3-12 to 3-26			0.559	-		
4-2	26,658	-	0.433	-	-793	-
4-9	10,760	-				
Abundance index $= \frac{(N_1 + N_2)}{2}$						18,709

Appendix IX. Weekly catch data used for determining population abundance of Niantic River winter flounder during 1980.

Week no.	Date-week of	Total catch	No. unmrk'd	No. marked	No. removed	No. exam'd	Recaps 1977-79	Recaptures (week marked) 1	2	3	4	5	Total recap
1	3-17	1,288	646	642	0	642	8	-					-
2	3-24	3,007	1,806	1,151	50	1,201	24	28	-				28
3	3-31	3,756	2,586	1,170	0	1,170	4	18	47	-			65
4	4-7	3,026	1,675	1,350	1	1,351	6	13	50	67	-		130
5	4-14	2,362	2,361	-	1	1,216	10	15	36	52	107	-	210
Total		13,439	9,074	4,313	52	5,580	52	74	133	119	107	-	433

Appendix X. The 1980 abundance estimate of winter flounder larger than 15 cm during the spawning period in the Niantic River.

Date-wk. of	Total number (N)	Standard error of N	Probability of survival (Φ)	Standard error of Φ	Calculated no. joining (B)	Standard error of B
3-17			0.664	0.102		
3-24	18,276	4,293	0.766	0.096	7,393	3,969
3-31	21,345	3,542	0.624	0.085	1,536	2,116
4-7	14,856	2,146				
Abundance index $= \frac{(N_1 + N_2 + N_3)}{3}$						18,159
± 2 standard errors $= \frac{2}{3}\sqrt{VarN_1 + VarN_2 + VarN_3}$						$\pm 3,976$

Appendix XI. Weekly catch data used for estimating population abundance of Niantic River winter flounder during 1981.

Week no.	Date-week of	Total catch	No. unmrk'd	No. marked	No. removed	No. exam'd	Recaps 1977-80	Recaptures (week marked) 1 2 3 4 5 6 7	Total recap
1	3-2	2,679	1,566	1,111	2	1,113	11	-	-
2	3-9	2,605	1,553	975	77	1,052	10	36 -	36
3	3-16	2,433	1,560	870	3	873	4	24 29 -	53
4	3-23	2,757	1,729	1,028	0	1,028	2	15 22 32 -	69
5	3-30	2,168	772	1,389	7	1,396	14	8 10 13 38 -	69
6	4-4	2,047	689	1,353	5	1,358	4	13 9 17 29 46 -	114
7	4-13	1,742	1,739	-	3	1,309	4	10 11 10 25 29 43 -	128
Total		16,431	9,608	6,726	97	8,129	49	106 81 72 92 75 43 -	469

Appendix XII. The 1981 abundance estimate of winter flounder larger than 15 cm during the spawning period in the Niantic River.

Date-wk. of	Total number (N)	Standard error of N	Probability of survival (Φ)	Standard error of Φ	Calculated no. joining (B)	Standard error of B
3-2			0.791	0.119		
3-9	25,674	5,636	0.681	0.109	2,955	4,122
3-16	20,378	3,932	0.583	0.091	5,963	2,747
3-23	17,842	3,068	1.097	0.171	28,288	7,214
3-30	47,858	8,622	0.757	0.150	-2,988	5,444
4-4	33,218	6,553				

Abundance index $= \frac{(N_2 + N_3 + N_4)}{3}$ 28,693

± 2 standard errors $= \frac{2}{3}\sqrt{VarN_2 + VarN_3 + VarN_4}$ $\pm 6,640$

Appendix XIII. Weekly catch data used for estimating population abundance of Niantic River winter flounder during 1982.

Week no.	Date-week of	Total catch	No. unmrk'd	No. marked	No. removed	No. exam'd	Recaps 1977-81	Recaptures (week marked) 1 2 3 4 5 6 7	Total recap
1	2-22	656	409	244	3	247	15	-	-
2	3-1	1,194	837	355	2	357	12	12 -	12
3	3-8	1,505	326	1,176	3	1,179	34	6 6 -	12
4	3-15	2,294	1,004	1,287	3	1,290	34	3 5 19 -	27
5	3-22	3,021	1,373	1,647	1	1,648	37	4 8 23 42 -	77
6	3-29	1,940	854	1,086	0	1,086	21	2 4 12 26 46 -	90
7	4-5	1,053	1,050	-	3	815	22	3 1 8 8 13 19 -	52
Total		11,663	5,853	5,795	15	6,622	175	30 24 62 76 59 19 -	270

Appendix XIV. The 1982 abundance estimate of winter flounder larger than 15 cm during the spawning period in the Niantic River.

Date-wk. of	Total number (N)	Standard error of N	Probability of survival (Φ)	Standard error of Φ	Calculated no. joining (B)	Standard error of B
2-22			1.213	0.335		
3-1	8,806	3,458	0.939	0.226	50,683	19,838
3-8	58,950	20,594	0.612	0.102	15,510	15,121
3-15	51,600	12,397	0.754	0.122	-1,142	8,971
3-22	37,768	6,868	0.578	0.110	2,510	3,772
3-29	24,340	4,896				

Abundance index $= \frac{(N_2 + N_3 + N_4)}{3}$ 49,439

± 2 standard errors $= \frac{2}{3}\sqrt{VarN_2 + VarN_3 + VarN_4}$ $\pm 16,666$

Appendix XV. Weekly catch data used for estimating population abundance of Niantic River winter flounder during 1983.

Week no.	Date-week of	Total catch	No. unmrk'd	No. marked	No. removed	No. exam'd	Recaps 1977-82	Recaptures (week marked) 1 2 3 4 5 6 7	Total recap
1	2-21	1,461	638	823	0	823	43	-	-
2	2-28	1,740	962	777	1	778	52	31 -	31
3	3-7	1,633	741	869	23	892	46	15 28 -	43
4	3-14	1,586	718	855	13	868	40	8 22 20 -	50
5	3-21	2,380	1,364	1,015	1	1,016	29	14 14 22 24 -	74
6	3-28	1,969	1,081	857	31	888	33	11 8 14 23 29 -	85
7	4-4	2,014	2,007	-	7	788	31	11 12 9 12 17 19 -	80
Total		12,783	7,511	5,196	76	6,053	274	90 84 65 59 46 19 -	363

Appendix XVI. The 1983 abundance estimate of winter flounder larger than 20 cm during the spawning period in the Niantic River.

Date-wk. of	Total number (N)	Standard error of N	Probability of survival (Φ)	Standard error of Φ	Calculated no. joining (B)	Standard error of B
2-21			0.701	0.108		
2-28	14,475	3,312	1.043	0.136	13,526	5,473
3-7	28,625	5,970	0.778	0.178	7,542	5,456
3-14	29,799	6,020	0.904	0.246	4,372	5,236
3-21	31,311	6,301			2,065	4,518
3-28	29,632	8,052				

Abundance index $= \frac{(N_2 + N_3 + N_4)}{3}$ 29,282

± 2 standard errors $= \frac{2}{3}\sqrt{VarN_2 + VarN_3 + VarN_4}$ $\pm 7,042$

Appendix XVII. Weekly catch data used for estimating population abundance of Niantic River winter flounder during 1984.

Week no.	Date-week of	Total catch	No. unmrk'd	No. marked	No. removed	No. exam'd	Recaps 1977-83	Recaptures (week marked) 1	2	3	4	5	6	7	8	Total recap
1	2-14	672	316	351	5	356	31	-								-
2	2-21	1,284	750	534	0	534	53	10	-							10
3	3-1	1,369	693	676	0	677	52	5	9	-						14
4	3-7	1,257	651	605	1	607	45	5	14	14	-					33
5	3-13	740	348	391	1	393	20	1	3	9	4	-				17
6	3-20	1,609	1,021	588	0	588	23	6	5	8	9	8	-			36
7	3-27	1,049	453	595	1	598	23	2	5	6	11	9	12	-		45
8	4-3	1,074	1,070	-	4	543	16	2	6	5	5	8	6	10	-	45
Total		9,054	5,302	3,740	12	4,296	263	31	42	42	29	25	18	10	-	197

Appendix XVIII. The 1984 abundance estimate of winter flounder larger than 20 cm during the spawning period in the Niantic River.

Date-wk. of	Total number (N)	Standard error of N	Probability of survival (Φ)	Standard error of Φ	Calculated no. joining (B)	Standard error of B
2-14			0.793	0.198		
2-21	14,955	5,904	0.997	0.217	23,912	12,380
3-1	38,815	12,801	0.849	0.206	-10,076	10,144
3-7	22,864	6,244	0.667	0.175	10,924	5,213
3-13 to 3-20	26,167	6,785	0.912	0.339	2,028	5,536
3-27	25,900	9,803				

Abundance index $= \frac{(N_2 + N_3 + N_4)}{3}$ 29,282

± 2 standard errors $= \frac{2}{3}\sqrt{VarN_2 + VarN_3 + VarN_4}$ $\pm 10,518$

Appendix XIX. Weekly catch data used for estimating population abundance of Niantic River winter flounder during 1985.

Week no.	Date-week of	Total catch	No. unmrk'd	No. marked	No. removed	No. exam'd	Recap. 1981-84	Recaptures (week marked)							Total recap
								1	2	3	4	5	6	7	
1	2-25	959	435	524	0	524	46	-							-
2	3-4	1,173	690	483	0	483	24	8	-						8
3	3-11	1,317	731	580	6	586	23	8	12	-					20
4	3-18	1,273	821	451	1	454	29	4	4	5	-				13
5	3-25	1,824	1.306	514	4	520	17	4	9	15	8	-			36
6	4-1	1,247	769	472	6	478	9	1	6	7	8	16	-		38
7	4-8	1,762	1,759	-	3	626	15	3	3	5	8	13	23	-	55
Total		9,555	6,511	3,024	20	3,671	163	28	34	32	24	29	23	-	170

Appendix XX. The 1985 abundance estimate of winter flounder larger than 20 cm during the spawning period in the Niantic River.

Date-wk. of	Total number (N)	Standard error of N	Probability of survival (Φ)	Standard error of Φ	Calculated no. joining (B)	Standard error of B
2-23			0.558	0.149		
3-4	17,637	7,676	0.829	0.210	4,014	6,963
3-11	18,642	5,912	0.843	0.225	19,518	11,286
3-18	35,236	12,685	0.527	0.147	-7,555	5,796
3-25	11,017	3,020	0.560	0.162	2,573	1,848
4-1	8,739	2,570				

Abundance index $= \frac{(N_2 + N_3 + N_4)}{3}$ 21,632

± 2 standard errors $= \frac{2}{3}\sqrt{VarN_2 + VarN_3 + VarN_4}$ $\pm 9,545$

Appendix XXI. Weekly catch data used for estimating population abundance of Niantic River winter flounder during 1986.

Week no.	Date-week of	Total catch	No. unmrk'd	No. marked	No. removed	No. exam'd	Recap. 1981-85	Recaptures (week marked)							Total recap
								1	2	3	4	5	6	7	
1	2-24	470	178	292	0	292	29	-							-
2	3-3	420	118	298	4	302	33	6	-						6
3	3-10	1,105	630	475	0	475	29	7	11	-					18
4	3-17	954	370	584	0	584	20	6	5	22	-				33
5	3-24	1,002	422	579	1	580	15	4	2	11	27	-			44
6	3-31	1,181	617	562	2	564	17	0	2	5	10	14	-		31
7	4-7	1,055	1,055	-	0	409	10	4	2	3	4	18	12	-	43
Total		6,187	3,390	2,790	7	3,206	153	27	22	41	41	32	12	-	175

Appendix XXII. The 1986 abundance estimate of winter flounder larger than 20 cm during the spawning period in the Niantic River.

Date-wk. of	Total number (N)	Standard error of N	Probability of survival (Φ)	Standard error of Φ	Calculated no. joining (B)	Standard error of B
2-24			0.995	0.286		
3-3	14,620	7,202	0.582	0.144	399	3,767
3-10	8,118	2,599	0.658	0.152	3,562	2,160
3-17	8,902	2,375	0.557	0.140	2,779	1,593
3-24	7,735	2,071	1.322	0.463	16,755	7,444
3-31	26,978	10,033				

Abundance index $= \frac{(N_2 + N_3 + N_4)}{3}$ 8,252

± 2 standard errors $= \frac{2}{3}\sqrt{VarN_2 + VarN_3 + VarN_4}$ $\pm 2,723$

Appendix XXIII. Number of tows by boat during Niantic River adult winter flounder abundance studies from 1976 through 1986.[a]

Year	Boat						
	Small craft[b]	Mya	Hamilton	Sisu	Sunbeam	Northeast I	Northeast II
1976	106	6	--	--	--	--	--
1977	154	--	--	--	--	--	--
1978	--	67	10	29	--	--	--
1979	--	40	2	51	--	--	--
1980	37	--	--	--	75	--	--
1981	13	--	--	--	--	57	39
1982	4	--	--	--	--	50	36
1983	--	--	--	--	--	63	72
1984	--	--	--	--	--	73	72
1985	--	--	--	--	--	76	80
1986	--	--	--	--	--	89	90

[a] During four-week period from mid-March through early April.
[b] Various outboard motor-powered boats of similar size.

Appendix XXIV. Number of tows per station during Niantic River adult winter flounder abundance studies from 1976 through 1986.

Station[a]	Year										
	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986
					Entire study						
1	92	285	159	145	166	100	143	56	49	42	39
2	67	63	88	38	30	43	50	21	36	41	36
3	27	3	0	4	--	--	--	--	--	--	--
4	20	3	6	30	20	3	12	23	31	32	39
5	14	2	4	17	1	137	117	--	--	--	--
6	75	41	14	14	11	3	--	--	--	--	--
7	41	7	16	14	--	--	--	--	--	--	--
8	17	5	2	3	--	--	--	--	--	--	--
51	--	--	--	--	--	--	--	64	78	58	58
52	--	--	--	--	--	--	--	33	38	48	70
53	--	--	--	--	--	--	--	36	49	48	62
54	--	--	--	--	--	--	--	--	--	--	11
					Comparative period[b]						
1	54	115	63	62	83	39	27	32	21	24	18
2	26	28	37	17	18	8	12	12	20	24	19
4	4	2	2	3	11	1	7	13	17	20	22
5	7	2	1	7	0	59	44	--	--	--	--
6	21	7	3	4	0	2	--	--	--	--	--
51	--	--	--	--	--	--	--	36	41	30	27
52	--	--	--	--	--	--	--	21	17	29	45
53	--	--	--	--	--	--	--	21	29	29	43
54	--	--	--	--	--	--	--	--	--	--	5

[a] Shallow-water stations 3, 6, 7, and 8 dropped after 1979 or 1981. Station 5 subdivided after 1982 into 51-53.

[b] Four-week period from mid-March through April used for calculation of annual CPUE. Excludes stations 3, 7, and 8 during 1976-79.